

UNIVERSITY OF CALGARY

Carbon Dynamics of Recolonized Cutover Minerotrophic Peatland: Implications for
Restoration

by

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A THESIS

SUBMITTED TO THE FACULTY OF GRADUATE STUDIES
IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE
DEGREE OF MASTER OF SCIENCE

DEPARTMENT OF GEOGRAPHY

CALGARY, ALBERTA

AUGUST, 2011

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UNIVERSITY OF CALGARY

FACULTY OF GRADUATE STUDIES

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ABSTRACT

In North America mulching of vacuum harvested sites combined with blocking of the drainage system is widely used for peatland restoration to accelerate *Sphagnum* moss establishment. However, peat extraction in fen peatlands or exposure of deeper minerotrophic peat layers results in soil chemistry that is less suitable for re-establishment of *Sphagnum* moss. In this situation restoration of plant species characteristic of minerotrophic peatlands is desirable to return the site to a carbon accumulating system. In these cases, it may be worthwhile to maintain spontaneously revegetating species as part of restoration if they provide desirable ecosystem functions. This study investigated the role of six spontaneously recolonizing vegetation communities for CO₂ and CH₄ exchange for two growing seasons (2008 and 2009) at an abandoned minerotrophic peatland in southeastern Quebec. The results were then compared with bare peat and adjacent natural fen vegetation. It also investigated the total carbon balance for vegetation communities, with and without literature estimates of winter time CO₂ and CH₄ fluxes and annual DOC/DIC leaching, as well as incorporated global warming potential (GWP) for greenhouse gas balance for 2009 growing season. The result shows that revegetation at the cutover site increased CH₄ and CO₂ flux in growing season above that of the natural site. The main controlling factors for CH₄ flux were vegetation volume and water table. However, the major controlling factors for maximum net ecosystem exchange for CO₂ (NEE_{max} measured at PAR > 1000 μmol m⁻² s⁻¹) were vegetation volume, and soil and air temperature. The study found communities dominated by *Carex aquatilis*, *Eriophorum vaginatum*, *Scirpus atrocinctus* and *Typha latifolia* at recolonized cutover site and hollow and shrub vegetation at natural site was a net sink of C from total C balance perspective. However, calculation based on GWP for greenhouse gases changed the

result for *Typha latifolia* because of its high CH₄ emission characteristics. Incorporation of previously reported winter time CO₂ and CH₄ fluxes and annual leaching for DOC/DIC shows that the C sink function will decrease by 36%, 38%, 42%, 30%, 32% and 75% for Car_aqu, Typ_lat, Sci_atr, Eri_vag, Nhol and Nshrub respectively. From GWP point of view this will decrease the green house gas sink (CO₂-e) functions for Car_aqu, Sci_atr, Eri_vag, Nhol and Nshrub by 25%, 22%, 17%, 16% and 38% respectively. Finally, considering all ecosystem functions this research suggested that *Carex aquatilis*, *Eriophorum vaginatum*, and *Scirpus atrocinctus* may be considered for inclusion in plans for minerotrophic peatland restoration.

ACKNOWLEDGEMENT

At first I would like to express my gratefulness to the Almighty because without His blessings it was not possible to complete this work. Then, I would like to express my heartfelt gratitude, sincere appreciation and profound regards to Dr. Maria Strack, Department of Geography, University of Calgary for her encouragement, passion, guidance and support from the beginning to the end of my study.

I am thankful to Jordanna Branham, Priyanka Srivastava, Y.C.A. Zuback, Ellen Widdup, J.R.P O'Brien, Evie Sarasas, Tyler Lister and Vicky Bérubé for their help with collection of field data and laboratory analysis. I am also thankful to Dr. Line Rochefort, NSERC Industrial Research Chair and Canadian Sphagnum Peat Moss Association for providing access to the study site and research funding.

Md. Sharif Mahmood

August 2011

DEDICATION

This work is dedicated to my wife whose love and dedication encouraged me in achieving my goals.

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PREFACE

This dissertation partially consists of a manuscript that has been published in an academic journal (Chapter 3). The text and figures from the manuscript is reprinted largely as it appears in the journal publication, but with changes to some of the language, which were made to make this dissertation a more cohesive document. Additional changes to the reference style and figures have also been made to ensure consistency with the dissertation format. While this paper was co-authored, I am the lead author of the studies and am responsible for partial data collection, all statistical analyses and writing the manuscript. All manuscript figures are reproduced with permission from the copyright holders.

Journal Publication

Mahmood, M.S., Strack, M., (2011). Methane dynamics of recolonized cutover minerotrophic peatland: Implications for restoration. *Ecological Engineering* 37, 1859 – 1868, doi:10.1016/j.ecoleng.2011.06.007.

CHAPTER ONE: INTRODUCTION AND BACKGROUND

1.1. Introduction

Peatlands can be found all over the world. Peatlands covers 5-8 % of the world's land and freshwater surface (IPCC, 2000). About 80% of the peatland area is situated in temperate-cold climates in the northern hemisphere, particularly in Russia, Canada and the USA; the remaining peatlands are found in tropical-subtropical climates, particularly in south-east Asia (Joosten, 2004; Wetland International, 2006). Peatlands represent the world's most important natural carbon (C) stores, containing over 550 million tons of C, or 30% of all land-based C (Wetlands International, 2008). Moreover, peatlands play important roles in the global cycling of C as they are net sinks of atmospheric carbon dioxide (CO₂) and a large source of atmospheric methane (CH₄) (Gorham, 1991). Despite covering less than 3% of the Earth's land surface, boreal and subarctic peatlands store between 270 and 370 Tg C (1 Tg=10¹² g) as peat (Turunen *et al.*, 2002), which would amount to 34–46% of the 796 Tg C currently held in the atmosphere as CO₂ (IPCC, 2007). Tropical peatland C stores are estimated to be around 52 Tg C, with very large uncertainties (8–258 Tg C) (Hooijer *et al.*, 2006). These large amounts of deposited C in the peatlands suggests these systems have been acting as sinks of atmospheric CO₂ for millennia; but they also show the potential for large CO₂ and CH₄ fluxes to the atmosphere or dissolved organic/inorganic carbon (DOC/DIC) release to rivers if peatland C stores were to be destabilized by global warming and changes in land use.

Peatland drainage and harvesting have increased over the last century due to the increased use of peat in the energy, agricultural, and horticultural sectors. The most important reason for mining peat in Canada is horticultural purposes. In Canada ~24 000 hectares of peatlands have been drained for peat extraction activities for horticultural use with 14 000 ha currently in production (Environment Canada, 2010). Another reason for peat disturbance in

Canada is extraction of oil sands. According to Price *et al.* (2010) “as part of the extraction process, overburden materials along with the terrestrial vegetation and surficial hydrologic features that overlie the oil sand across hundreds of square kilometres of the land surface have been stripped off and stockpiled or used as construction materials”.

The impact of horticultural peat harvesting on the peatland ecosystem are severe. During peat harvesting bogs are drained, vegetation is removed and a thick layer of soil is harvested, usually occurring over a period of several decades (Waddington and Price, 2000). After cessation of peat harvesting the remaining peat surfaces are net sources of C to the atmosphere as the residual peat gradually decomposes (Tuittila *et al.*, 1999; Waddington *et al.*, 2002). Warner (1999) found that the net CO₂ loss to the atmosphere increased 300% at a mined peatland.

The restoration of abandoned cutover peatlands has recently been encouraged as a land-use management practice especially to reduce atmospheric CO₂ emissions. Waddington and Warner (2001) found that the net flux of CO₂ to the atmosphere decreased by over 100 g CO₂ m⁻² day⁻¹ at restoration plots. Waddington *et al.* (2010) determined that a restored peatland can become a net sink during the growing season within five years post restoration. Although these results are very promising for ecologists and land managers, more research is still needed to clarify the long-term successional trend at restored sites as only little is known about the long-term C dynamics related to peatland restoration (Yli-Petays *et al.*, 2007).

Bogs are a type of peatland which receives water solely from rain and/or snow falling to its surface. On the other hand, fens are peatlands which are influenced by both water from precipitation and below ground (Charman, 2002). In North America, peatland restoration, mostly focused on bogs, involves introduction of diaspores and mulching of harvested sites with blocking the drainage to accelerate *Sphagnum* establishment (Rocheffort *et al.*, 2003).

Restoring only the hydrology of the peatland is not always sufficient (Yli-Petays *et al.*, 2007) as the environmental conditions, especially the soil chemistry is completely changed, a seed bank is lacking (Campbell *et al.*, 2003), and water table and temperature fluctuations may be very high (van Seters and Price, 2002). Furthermore, in fens, establishment of *Sphagnum* may not be the goal due to hydro-chemical conditions. Establishment of vascular vegetation following harvesting is also generally more extensive than that observed on disturbed bogs (Graf *et al.*, 2008). Restoration of fens has been limited, due to lack of knowledge regarding target vegetation communities for re-establishment, and the impacts of revegetated species on the C flux rate. Thus, in fens, spontaneously recolonizing species resulting from natural succession could play an important role in ecosystem recovery if they have the potential to act as C sink, as they are more stable and cost less than active, imposed restoration strategies (Bradshaw, 2000; Parch *et al.*, 2001).

1.2. Objectives

The overall objective of my research was to determine whether spontaneously recolonizing species of cutover minerotrophic peatlands (fens) could be included in restoration from the carbon perspective. Therefore my specific research objectives were:

- I. To measure the carbon flux of representative species at an abandoned mined peatland;
- II. To compare the carbon flux of these species with that of bare areas and natural peatland to determine the effect of mining and revegetation on carbon fluxes;
- III. To investigate controls on CO₂ and CH₄ flux at the natural and cutover peatland to assist in the development of restoration strategies to maximize C storage; and
- IV. To identify spontaneously recolonizing species with the potential for carbon accumulation.

1.3. Literature Review

1.3.1. Peat and Peatlands

Peat is undecomposed plant material that has accumulated over thousands of years. Due to the absence of oxygen in water-saturated environments, the decomposition of this material has been slowed. It is possible to define it in two ways. Firstly, which is most common, peat is defined as a substance that is composed of the partially decomposed remains of plants with over 65% organic matter (dry weight basis) and less than 20-35% inorganic content (Clymo, 1983). It is also possible to define peat in its intact or natural state (i.e. in the peatland) as 88-97% water, 2-10% dry matter and 1-7% gas (Heathwaite *et al.*, 1993). According to Charman (2002) “the term ‘peatland’ has different meanings to different people in many disciplines, which led to confusion and misunderstanding of the nature of peat and peatlands”. Charman (2002) also introduces some common key terms that are usually used in peatland literature that are presented in Table – 1.1.

Glaser (1987) defined peatlands as any area with a waterlogged predominantly organic substrate of at least 30–40 cm thickness. In Europe and Russia, there is a tendency to differentiate actively peat-accumulating *mires* from the more generic *peatland* term, which applies to all peatland regardless of whether or not the system is actively accumulating peat (Charman, 2002; Table -1.1). According to Warner (2001) the term ‘mire’ is not extensively used in North America, and as an alternative the more generic ‘peatland’ term is used in the broadest sense to denote any peat landform irrespective of whether it is two-layered and actively forming and accumulating peat or not. According to Canadian System of Soil Classification (1998), peatlands are soils that are composed predominantly of organic matter (more than 30% of organic matter by weight), in the upper half meter. However throughout this thesis I will use three terms as three different systems i.e. a) cutover site or peatland for

harvested site where recolonization occurred by studied vegetation communities, b) bare peat for harvested site where recolonization of vegetation communities is lacking, and c) natural site or peatland for undisturbed part surrounding the cutover site.

1.3.2. Structure of Peatlands

Peatlands are both horizontally and vertically structured (Blodau, 2002). Vertical peat profiles consist of a saturated zone, in which oxygen is depleted and organic matter is anoxically decomposed (“catotelm”) (Blodau, 2002). A surface layer (“acrotelm”), on top of the catotelm can be found (Clymo, 1984; Table –1.2). This layer is often oxic but can seasonally be saturated and anoxic. According to Charman (2002) acrotelm is known as the active layer, and it is the zone in which most growth and decay occur, and in which living organisms mostly exist. The acrotelm consists of a productive layer, a litter layer, and a collapse layer in which the macrostructure of the plants is lost (Warner, 1996). Acrotelm and catotelm store approximately 98.5% of the total C in peatlands (Gorham, 1991). Carbon stored in vegetation contributes on average only about 1.5% ($\sim 2000 \text{ g m}^{-2}$) to the C pool, but this contribution may vary substantially (350 to 6000 g m^{-2} ; Grigal *et al.*, 1985).

Horizontally, peatlands are structured on different scales. “Microforms” are developed mainly within the vegetation layer and the acrotelm. A typical example is the hummocks and hollows or lawns, which represent elevated and indented areas respectively and display distinct vegetation associations (Rocheftort *et al.*, 1990; Warner 1996). The latter group is composed of habitats that form depressions where the water level is close to the surface. Depending on the area they cover these habitats are called lawns or hollows. Lawns cover large surfaces, while hollows are small depressions (Quinty and Rocheftort, 2003).

1.3.3. Classification of Peatlands

Peatlands have been classified based on several criteria e.g., vegetation characteristics, geomorphology, hydrology, chemistry, stratigraphy, and peat characteristics, or on combinations of these criteria (Blodau, 2002). This has resulted in complex classification systems (Moore and Bellamy, 1974; Mitsch and Gosselink, 1993). According to Davis and Anderson (2001) peatlands are classified as two major types; bogs and fens. In North America four basic types, ombrotrophic bogs, minerotrophic fens, intermediate or “poor” fens, and calcareous fens are distinguished (Blodau, 2002). These types differ with respect to pH, base cation concentrations, dominating vegetation, and hydrologic dynamics (Glaser, 1987; Vitt *et al.*, 1995; Mullen *et al.*, 2000). Detailed descriptions of peatland classification can be found in Charman (2002).

Ombrotrophic Bogs

Bogs are peatlands with a convex shape such that the center is raised as much as several meters above the edge (Reeve, 2010). This profile does not allow water influenced by mineral soil to reach the raised portion of the bog. Bogs are *ombrotrophic* meaning that mineral nutrients received by the plants come entirely from the atmosphere in wet (precipitation) and dry fallout (Davis and Anderson, 2001) and not from other mineral soil sources. Many of the plants that form the bog are uniquely suited to get their nutrients in this way. The term “bog” refers to only the unit of the peatland that is raised above the rest. In a peatland complex water from mineral soils affects the edges of the peatland. This area is known as the fen or lag (Davis and Anderson, 2001). For this reason, peatlands that contain bogs are often surrounded by fens.

Minerotrophic Peatlands/Fens

There is a vast array of minerotrophic peatland or fen types throughout the world and it is difficult to provide a comprehensive summary of terminology or to describe a system that adequately caters for all of this variation (Charman, 2002). According to Davis and Anderson (2001) fens are *minerotrophic*, meaning that the plants growing on fen surfaces receive some of their mineral nutrients in water that has been in contact with underlying or surrounding mineral substrates. Minerotrophic fens are influenced by more alkaline, nutrient-rich groundwater and are dominated by sedges, herbs and bryophytes (Lai, 2009). Fens can be *oligotrophic* (nutrient poor) or *eutrophic* (nutrient rich), or intermediate, depending on hydrology and local geology/mineralogy (Davis and Anderson, 2001). Fens come in many varieties including ribbed, and pattern less fens. Fens are far more common than bogs, because they form virtually anywhere water is allowed to pool or run slowly across a surface (Reeve, 2010). Fens support a much greater range of vegetation than bogs, because of the availability of certain nutrients. Fens form on the edges of a bog where there is mineral soil influence. Fens can also make up an entire peatland. Aapa mires, for example, are a type of northern peatland that is dominated by fen (Reeve, 2010).

1.4. Peatlands and Carbon Cycling

The main source of C stored in peatlands is atmospheric CO₂. The other source of peatland C is in dissolved form, i.e. Dissolved Organic Carbon (DOC) and Dissolved Inorganic Carbon (DIC) which come from both surface and ground water. The plants uptake CO₂ from the atmosphere by photosynthesis during the growing season and afterwards C is deposited as litter both on and in the soil (Vasander and Kettunen, 2006). As the water table lies near the soil surface, which leads to anoxic conditions, decay rates of this litter are low (Kivimäki *et al.*, 2008). According to Vasander and Kettunen (2006) peat accumulates

whenever the rate of organic matter production exceeds the rate of decay. The “efficiency” of peatlands, which is the ratio between peat accumulation and Net Primary Productivity (NPP), varies between 1 and 20 % (Moore *et al.*, 2002; Feng, 2002) although the NPP and peat accumulation values in different kinds of peatlands differ widely. The peat accumulation rate has been related to peatland geographical location (south greater than north), age (young greater than old), and type (Vasander and Kettunen, 2006).

There are several ways by which the fixed C of peatlands is lost from the system. These are – 1) respiration by the plants themselves to maintain growth of above ground and below ground plant parts and by consumers such as soil animals and heterotrophic microbial communities, 2) methane (CH₄) emission, and 3) as DOC and DIC in discharge. By measuring the total efflux of CO₂ it is impossible to distinguish respiration by plants and microbes from each other, but by other means it has been found that the combination of autotrophic and heterotrophic respiration comprises about one third of the CO₂ uptake via photosynthesis during the intensive growth period (Bubier *et al.*, 1998; Heikkinen *et al.*, 2002).

1.4.1. Controlling Factors of CO₂ Exchange

Rates of photosynthesis and respiration, which combine to give a value for net ecosystem exchange (NEE), are of fundamental importance. Simply, if the NEE is negative then the system is acting as a sink for CO₂ but if it is positive, then the peatland is a source (Charman, 2002). The process of photosynthesis is light dependent in which CO₂ is the C source and light is used as energy (Vasander and Kettunen, 2006). Mooney (1986) found CO₂ concentration, temperature, water and nutrient availability as well as the leaf area also affect photosynthesis. According to Vasander and Kettunen (2006) when photosynthetic active radiation (PAR) increases to about 200 $\mu\text{mol m}^{-2} \text{s}^{-1}$ rates of photosynthesis equal rates of

respiration ($NEE = 0$), and with increasing PAR peatlands become net CO_2 sinks. The results of Kettunen (2000) showed that CO_2 exchange is extremely sensitive to variation in environmental factors on a short-term time scale and, consequently, annual C exchange estimates are also affected by short-term variation (Bubier *et al.*, 1998, 1999; Griffis *et al.*, 2000; Soegaard *et al.*, 2000). Long day length and moderate temperatures allow high growth rates in the early summer, but as the season progresses, respiration increases and photosynthesis falls so that the C balance is positive for some periods from late July onwards (Charman, 2002). According to Shurpali *et al.* (1995) high temperatures and relatively low water table kept photosynthesis low and emission of CO_2 relatively high, making the peatland under study a net source of CO_2 to the atmosphere in this situation. Recent studies of Lund *et al.* (2010) on 12 northern peatlands and tundra sites found that leaf area index (LAI) has significant positive relationship with NEE i.e. more LAI would increase C uptake.

The productivity of the vegetation is related to the vegetation community of peatlands and is driven by the nutrient status, hydrology (Malmer, 1986) and peat temperature (Shurpali *et al.*, 1995). Frolking *et al.* (1998) found from a variety of northern peatlands that at high light levels NEE was greater at rich fens and poor fens than at bogs. Different taxa have different optimal hydrological conditions for photosynthesis (Wallen *et al.*, 1988) as well as environmental, for example, Silvola and Aaltonen (1984) found when moisture drops below optimal levels in *Sphagnum*, the photosynthesis rate declines quickly while excess moisture had a much smaller effect. *Sphagnum* moss can tolerate very wet, acidic and nutrient poor conditions (Clymo and Hayward, 1982) and can photosynthesize efficiently at low temperatures (Harley *et al.*, 1989). However, sedges (for example, *Carex* spp., *Eriophorum* spp.) can tolerate anoxic, waterlogged conditions and are strong competitors in such habitats

as they are perennial and deep rooting, with the ability to transport oxygen from the above ground parts to the roots via aerenchymatous tissue (Riutta *et al.*, 2007).

The rate of autotrophic respiration is regulated by photosynthesis, temperature, and water and nutrient availability, while heterotrophic respiration is controlled largely by soil temperature, oxic peat layer volume, nutrient availability, soil pH, and the quality and quantity of decomposable material (Chapman and Thurlow, 1998; Chapin *et al.*, 2002). Root-associated respiration follows the vegetation phenology and may account for 10–45 % of the total soil CO₂ release (Silvola *et al.*, 1996a), originating mainly from the turnover of fine roots and from root exudates. The remaining C is transformed into plant structures, especially into the belowground parts of plants but not for all peatland species. In the water-saturated anoxic part of the peat (catotelm), decomposition is slow and a large portion of the total mineralized C is released to the atmosphere as CH₄ (Vasander and Kettunen, 2006). Heterotrophic (microbial) respiration rates are very high when there is a deeper water table as the decomposition rate is faster under oxic conditions compared to anoxic conditions (Moore and Dalva, 1993). However, Lafleur *et al.* (2005) concluded that as respiration is an enzymatically-controlled process, growing season ecosystem respiration in a dry ombrotrophic bog is strongly correlated with peat temperature and very weakly correlated with water table depth. Updegraff *et al.* (2001) showed that respiration rates increase with temperature but not to a variation in water-table. Bubier *et al.* (1998) concluded for a number of boreal peatland types that trophic status, above-ground net primary productivity, and soil temperature were more important than water table in determining ecosystem respiration. But, recent findings of Lund *et al.* (2010) showed that respiration was lower than productivity in 12 northern peatland and tundra sites as high water table suppressed decomposition rates. The type of vegetation community is also related to respiration rate because of differences in autotrophic respiration

and because decomposability of organic matter and peat substrates varies among plant species (Moore and Basiliko, 2006). Moreover, the chemistry or quality of the substrate available affects respiration. As new litter is decomposed the remaining substrate becomes increasingly recalcitrant and more difficult for microbes to degrade and is manifested in lower decomposition rates (Strack *et al.*, 2008).

1.4.2. Methane Dynamics and Peatlands

Methane (CH₄) is a greenhouse gas with 25 times the global warming potential of CO₂ on a 100-year time scale (IPCC, 2007). CH₄ concentrations have been increasing by 7.0 part per billion (ppb) per year from pre-industrial revolution concentrations. For example, 1998 concentrations of 1745 ppb were 250% higher than pre-industrial revolution concentrations (IPCC, 2007). Although the atmospheric concentration of CH₄ is 1/200 of CO₂, it accounts for 20% of the radiative forcing of all greenhouse gases in the atmosphere (IPCC, 2007). CH₄ is particularly important in the global climate change debate because it has a mean residence time in the atmosphere of 12 years (IPCC, 2007), magnifying the radiative forcing of CH₄ and causing a higher global warming potential from CH₄ than from CO₂ in the short term. CH₄ is destroyed in a reaction with OH⁻ radicals, forming water and potentially ozone (Tyler, 1991). CH₄ has a significant effect on the chemical interactions of the Earth's atmosphere, such as through the formation of ozone, another important greenhouse gas, and consequently has the potential to enhance global climate change both through its own global warming potential and through the formation of other greenhouse gases. CH₄ is produced both anthropogenically and naturally. Anthropogenic sources of CH₄ account for 60% of methane emissions while natural sources produce the rest (IPCC, 2001). Of both anthropogenic and natural sources, wetlands comprise the largest single source of CH₄ to the atmosphere (IPCC, 2001).

Methane Production

Methane (CH₄) is a trace gas that is the end product of a long string of reactions that break down organic matter in anoxic environments. CH₄ formation occurs through two different pathways, bicarbonate reduction/hydrogen pathways and acetate fermentation. Both processes are carried out by methanogenic archaea in the absence of oxygen and other alternate electron receptors (Westermann and Ahring, 1987).

The hydrogen pathway and the acetate pathway differ in their importance to CH₄ production. The reduction of CO₂ via the hydrogen pathway to form CH₄ becomes more important with peat depth as labile C is less available (Popp *et al.*, 1999), but is relatively unimportant in most peatlands with high amount of labile C. Where labile C is readily available, acetate fermentation is the dominant pathway of CH₄ formation in many peatlands. Acetate fermentation accounts for 85-90% of CH₄ produced in ecosystems with low temperatures (Avery *et al.*, 1999). Acetate is formed from labile C in the peat, from root exudates, organic matter, and recent photosynthates from plant roots (King and Reeburgh, 2002).

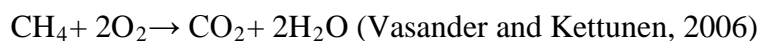
Older reviews concluded that much of the substrate for methanogenesis was derived from peat decomposition (Panikov, 1999), but more recent research has shown that recently fixed C is an important source of substrate for methanogens. King and Reeburgh (2002) found a fraction of isotopically labelled carbon taken up as CO₂ during photosynthesis is released as CH₄ as soon as two hours after uptake, as vascular plants allocate recently fixed C to their roots. Similarly, isotopically labelled acetate was emitted as CH₄ at the highest rates between 72 and 240 hours after addition (Strom *et al.*, 2003), indicating that labile C is used very quickly. The rapid turnover of the C substrates to CH₄ indicates the importance of recently fixed C to CH₄ production. Substrate availability is a key determinant of CH₄ production

(Strom *et al.*, 2005). The analysis of methanogenic communities from boreal mires supported the idea that upper peat layers that receive fresh organic matter harbour acetoclastic methanogens, while hydrogen-utilizing methanogens prevail in deeper layers (Galand *et al.*, 2003).

Methanogens are active only under anoxic conditions, so the production of CH₄ is dependent on the water table level. Although methanogens can survive dry periods with deep water table levels, they generally do not produce any CH₄ during these periods (Blodau and Moore, 2003; Moore and Dalva, 1993).

Methane Oxidation

CH₄ fluxes are the net result of CH₄ production in the anoxic zone of the peat, and CH₄ oxidation in the oxic peat layers. In wetlands, CH₄ oxidation is carried out by low-affinity CH₄ oxidizers and oxidation rates depend on CH₄ and oxygen availability, which are connected to peat moisture conditions, temperature, and the activity of CH₄ oxidizing bacteria in the peat matrix (Vasander and Kettunen, 2006). CH₄ oxidation or methanotrophy occurs in oxic environments as anoxically produced methane is oxidized by methanotrophic bacteria to form CO₂ in the following equation.



CH₄ oxidation has a significant impact on CH₄ fluxes. Oxidation may consume up to 90% of CH₄ produced in a peatland (King, 1990a). The proportion of CH₄ oxidized is dependent on the concentration of CH₄ in the peat (Moosavi and Crill, 1998; Sundh *et al.*, 1995), which is dependent on production. However, CH₄ can also be transported through other mechanisms and bypass oxidation, increasing the net efflux from the system. Temperature control has been suggested to be less important for CH₄ oxidation than for CH₄ production (Dunfield *et al.*, 1993).

Methane Transport

The CH₄ flux that is measured at the surface is a combination of three different transport mechanisms, diffusion through the peat, ebullition or bubble release, and plant mediated transport (Vasander and Kettunen, 2006). Diffusion contributes a much smaller proportion to CH₄ fluxes than plant mediated transport and ebullition (Chasar *et al.*, 2000; Dove *et al.*, 1999), but this is dependent on the environmental conditions and vegetation within a peatland. At unvegetated surfaces, ebullition mainly dominates (van der Nat *et al.*, 1998). In vegetated surfaces, bubble release may become important during winter time when plant biomass is low (van der Nat and Middelburg, 1998).

CH₄ diffusion through the peat is a passive physical process. CH₄ travels from the anoxic zone of the peat at or below water table and through the oxic zone of the peat along concentration gradients. The CH₄ efflux is significantly less than the CH₄ that is produced due to CH₄ oxidation in the oxic zone. The relative importance of these two processes is controlled by the water table. Areas with a deep water table level both produce less CH₄ due to the reduced size of the anoxic zone (and presumably less labile C), and oxidize more CH₄ due to the increased oxic zone, resulting in a lower net CH₄ flux (Lai, 2009).

Ebullition can contribute significantly to CH₄ fluxes. Ebullition or bubbling of gases could happen anywhere. Ebullition and bubble release are significant contributors to annual CH₄ fluxes because they are emitted as pulses that are too large to be fully oxidized, unlike CH₄ diffusion through the peat. Christensen *et al.* (2003a) estimated that ebullition fluxes accounted for 18-50% of total CH₄ emissions. A possible cause of episodic fluxes is a drop in water table level (Windsor *et al.*, 1992) or a change in atmospheric pressure (Bubier *et al.*, 1993; Moore and Dalva, 1993; Scranton *et al.*, 1993, Tokida *et al.*, 2005, Strack *et al.*, 2006), which causes dissolved CH₄ in the peat to be released.

Plant mediated transport of methane, the final mechanism by which CH₄ is released to the atmosphere, has been found to account for between 48% and 97% of CH₄ effluxes (Christensen *et al.*, 2003b; Kelker and Chanton, 1997; Shannon *et al.*, 1996). Vascular plant species have differing effects on CH₄ emissions depending on whether a plant is aerenchymatous or not (Shannon and White, 1994). Vascular plants with aerenchymatous tissue, like sedges, have passages that allow both oxygen to travel to their roots and CH₄ to escape to the atmosphere. Ericaceous shrub such as leather leaf (*Chamadaphne calyculata*) does not have this type of system. Plants transport oxygen to their roots for root respiration, which also enables CH₄ oxidation when dissolved CH₄ in the peat is exposed to oxygen (Bellisario *et al.*, 1999; Shannon *et al.*, 1996). However, by transporting a gas to their roots, plants also enable CH₄ to diffuse through the plant roots and bypass oxidation in the peat. Additionally, vascular plants allocate C to their roots, increasing the substrate available for methanogenesis. Several studies have shown that sites with vascular vegetation have much higher CH₄ emissions than sites where vascular vegetation has been removed or clipped (Christensen *et al.*, 2003b; Kelker and Chanton, 1997; Saarnio *et al.*, 1998; Waddington *et al.*, 1996). The strength of vascular plant control, however, is determined by species composition and the water table level. Wetland sites with sedges have been found to have the highest CH₄ fluxes (King *et al.*, 1998), 6 to 12 times higher than *Sphagnum* sites (Saarnio *et al.*, 1998) and significantly higher than sites with Ericaceous shrubs (Shannon and White, 1994). Sedges are aerenchymatous plants, and it has been well documented that the presence of sedges leads to higher CH₄ fluxes due to their physical properties (Kelker and Chanton, 1997). However, Waddington *et al.* (1996) suggest that the influence of vascular plants on CH₄ fluxes is correlated with the water table level. They suggest that CH₄ fluxes may only be enhanced by vascular vegetation when the water table is deep, because the plant roots are more likely to

reach the zone of CH₄ production, thus contributing substrate through root exudates and enabling transport from the anoxic zone of CH₄ production.

1.4.3. Controls on Methane Flux

The CH₄ fluxes from wetlands, which are controlled by the dynamic balance between CH₄ production and oxidation rates in peat profiles and the transport rate from peat to the atmosphere (Conrad, 1989), show high spatial and temporal variation (Kettunen, 2003). The factors controlling CH₄ fluxes are mainly vegetation, water table and temperature although other factors such as atmospheric pressure, storms and microbial communities have significant effect.

Vegetation

Vegetation influences CH₄ fluxes through substrate production and through plant-mediated transport. The presence of vascular vegetation has been found to increase CH₄ emissions (Shannon and White, 1994; Shannon *et al.*, 1996; Waddington *et al.*, 1996), despite the associations described between active CH₄ oxidizing bacteria and the rhizospheres of sedge species (Popp *et al.*, 2000) and plant fine root material (Gerard and Chanton, 1993). CH₄ oxidation is not as tightly coupled to vascular plants as CH₄ production. Many studies have linked plant productivity and C assimilation to CH₄ fluxes (Joabsson and Christensen, 2001; King *et al.*, 2002; Whiting and Chanton, 1993; Whiting *et al.*, 1991). This is probably due to C allocation to the plant roots and subsequent plant root exudates, which leads to increased substrate for methanogenesis (Joabsson *et al.*, 1999a; King *et al.*, 2002; Ström *et al.*, 2003). Other research has shown that CH₄ fluxes are related to light-regulated root exudation (Mikkela *et al.*, 1995) although this might also be a function of light-related oxidation processes, which are inhibited by light (King, 1990b). Van den Pol-van Dasselaar *et al.* (1999) found that plant residues in the peat surface (from 0-5cm depth) contributed to 70% of the

total CH₄ production capacity of the peat, indicating that plants are crucial to the formation of substrate for methanogenesis.

The plant species composition of a wetland also affects the CH₄ fluxes through the amount of CH₄ transportation and the substrate production (Table – 1.3). Ström *et al.* (2003) found that different vegetation species had different rates of acetate production.

Studies have found varying qualities of labile C in the peat, depending on the dominant species type (Aerts and Toet, 1997; Bergman *et al.*, 1998; Valentine *et al.*, 1994). Shrub dominated areas tend to have recalcitrant C sources within the peat, while the C found in sedge dominated areas of the peat is more labile (Yavitt and Lang, 1990).

Water Table

Water table level controls the depth of the anoxic layer below the peat surface and therefore controls the size of the zone of CH₄ production and CH₄ oxidation. Higher CH₄ emissions have been observed with a shallower water table level, while lower emissions have been observed at deep water table levels (Wilson *et al.*, 2008; Aerts and Ludwig, 1997; Bubier *et al.*, 1993; Dise *et al.*, 1993; Moore and Roulet, 1993). Several researchers have found that the water table is a major control of CH₄ emissions and found that the seasonal average water table may be the best predictor of CH₄ emissions (Wilson *et al.*, 2008; Bubier *et al.*, 1993; Bubier, 1995; Dise *et al.*, 1993; Moore and Dalva, 1997; Waddington *et al.*, 1996).

However, the relationship between water table and CH₄ emissions is not always direct. An inverse relationship from that which is expected and described above has been measured in several studies. High CH₄ fluxes during a period of shallow water table position have been observed when the average water table was within 15 cm of the peat surface (Bellisario *et al.*, 1999; Kettunen *et al.*, 1996). Insignificant correlations between CH₄ fluxes and water table level with small variations in the water table have been found (Moosavi and Crill, 1997;

Shannon and White, 1994; Yavitt *et al.*, 1993). Van den Pol-Van Dasselaar *et al.* (1999) found a 5 cm change in the water table could decrease or increase CH₄ emissions by 30-50%. Furthermore, Kettunen *et al.* (2000) found that water table position did not explain any additional variability of CH₄ fluxes than that accounted for by peat temperature. Frohling and Crill (1994) also found that the relationships between water table level and CH₄ fluxes did not hold true for all years of their 3-year study, depending on yearly environmental variations. These findings indicate that although the relationship between water table level and CH₄ emissions is generally positive, it does not hold true in all cases, especially when there are small variations in water table level or it is near the peat surface.

The relationship between CH₄ effluxes and water table level becomes less certain when the water table level experiences small fluctuations or is close to the peat surface. It is possible that with small changes in water table or with a deep water table, CH₄ fluxes become decoupled from the environmental factors. Instead fluxes become dependent on plant activities, such as plant transport and substrate production, leading to the observed correlations between plant productivity and CH₄ emissions. With small changes in water table, these processes may vary in importance. A possible explanation for the observed high emissions during periods of shallow water table is the efflux of CH₄ due to a pressure gradient between the zones of production and the atmosphere (Kettunen *et al.*, 1996). Both departures from the hypothetical relationship of CH₄ to water table indicate that the relationships between CH₄ fluxes water table are unimodal and scale dependent.

Temperature

Peat temperature is one of the most important controlling factors for CH₄ emissions. CH₄ production is a microbially mitigated process, so like all microbial processes; temperature controls the rate of activity. The correlation between peat temperature and CH₄

production has been described by many researchers (Bergman *et al.*, 2000; Christensen *et al.*, 2003a; Dise *et al.*, 1993; Frolking and Crill, 1994; Kettunen *et al.*, 2000; Shannon and White, 1994; Updegraff *et al.*, 2001; van Hulzen *et al.*, 1999). Several studies have described seasonal patterns of CH₄ fluxes (Waddington *et al.*, 1996; Windsor *et al.*, 1992) are smaller in the beginning of the season due to the lower peat temperature. As the peat warms throughout the summer season, CH₄ fluxes increase in magnitude. In the winter, CH₄ fluxes are 10% to 100% of their summer values (Heikkinen *et al.*, 2002).

Air temperature and water table level control peat temperature. The lag time between changes in air temperature and peat temperature is a likely cause of the observed seasonal patterns of CH₄ emissions (Frolking and Crill, 1994). Peat temperature has been shown to account for most of the variability in CH₄ fluxes, especially when used with water table data, and the seasonal mean of both water table and peat temperature has been used to predict CH₄ emissions (Bubier *et al.*, 1995). The effects of peat temperature on CH₄ emissions are also somewhat dependent on the vegetation of a given site. Sites dominated by vascular, aerenchymatous plants have a stronger positive relationship between CH₄ fluxes and peat temperature at depth than other sites (Saarnio *et al.*, 1998; Shannon and White, 1994).

1.5. Peatland Restoration

Peatland restoration comprises all deliberate action that initiates or accelerates the recovery of a degraded peatland to a former better state (Schumann and Joosten, 2008). Peatland restoration is a relatively new field of investigation that has seen significant advances in the 1990s (Rocheffort *et al.*, 2003). The first restoration attempts of cutover peatlands, carried out in Germany, the United Kingdom and the Netherlands, included the rewetting of peat deposits by filling or blocking drainage ditches (Tuittila *et al.*, 2000c). The restoration technique in Europe is mainly limited to rewetting of cutover sites (Vasander *et al.*, 2003; Yli-

Petäys *et al.*, 2007). However, because of different land uses of peatlands, peat mining methodology, and goals for restoring regional biodiversity (Rochefort *et al.*, 2003) the European and North American approaches to peatland restoration differ greatly (Money and Wheeler, 1999). The North American peatland restoration method consists of the following steps: 1) field preparation, 2) diaspore collection, 3) introduction and protection of diaspores, and 4) fertilization (Rochefort *et al.*, 2003). Diaspores are any part of a plant that can regenerate a new individual such as seeds, rhizomes, shoots or branches (Rochefort *et al.*, 2003).

The intention of peatland restoration is to bring back a naturally functioning, self-sustaining ecosystem (Wheeler and Shaw 1995), or reinitiate the peat forming processes (Pfadenhauer and Klötzli 1996). According to Yli-Petäys *et al.* (2007) the aim of restoration is to promote peatland vegetation establishment by altering factors hindering plant colonization and survival. Rochefort (2000) defines the goals of peatland restoration as re-establishing 1) a plant cover dominated by *Sphagnum* or brown mosses, depending on the status of the residual peat and 2) the diplotelmic hydrological layers (i.e. the acrotelm and catotelm) that characterize intact 'active' peatlands. According to Rochefort (2000) "Peatland restoration also implies the return of a functionality that ensures ecosystem maintenance in the long term e.g., achieving an adequate level of productivity, returning the mined site to a peat accumulating system, re-establishing the cycling of nutrients, returning a vegetation structure and microhabitats from which emerge faunal and floral diversity, and making sure that the ecosystem is resistant to biological invasion". Gorham and Rochefort (2003) suggested that if the damage to a degraded peatland is not too severe the primary goal is to repair the ecosystem, or to rebuild it if there has been considerable loss of peat. This will involve re-establishing more or less normal peatland hydrology, biogeochemical cycling, and energy

capture that will allow autogenic plant succession (Glaser and Janssens, 1986; Foster and Wright, 1990) and the renewal of peat accumulation. Once harvesting has been completed it is important to act as soon as possible in order to minimize degradation of the surface peat by decomposition and compaction (Schouwenaars, 1993); further losses by wind or water erosion, frost heaving and ice formation (Quinty and Rochefort, 2000) and potentially return the C sink function of the ecosystem (Gorham and Rochefort, 2003). Otherwise, recolonization by plants can be negatively affected due to altered hydrology of the peatland and wind erosion or frost heaving damage (Poulin *et al.*, 2005).

The short-term objectives of present peatland restoration are to establish plant cover composed of peat bog species (3 to 5 years) with particular attention to *Sphagnum*, and to recreate hydrological conditions similar to natural bogs (Rochefort *et al.*, 2003). However, disturbance may often lead to irreversible changes at the cutover site which can make it difficult to re-establish the conditions that are essential for the formation and growth of peatland (Höper *et al.*, 2008). Other factors such as climate change and eutrophication could impede or reduce peat formation at the local or global level. Moreover peat growth alone is not sufficient as an indicator to assess the net climatic effect of peatland restoration, as the exchange of climatically relevant trace gases (e.g. CO₂, CH₄) determines the climate effect (Höper *et al.*, 2007). Achieving these goals is more difficult for fen restoration as very little research has been conducted on restoring fen peatlands.

A post-mined peatland is a harsh environment for plants (Lavoie *et al.*, 2003). Several studies have characterized the spontaneous colonization of harvested peatlands with ombrotrophic residual peat (harvested bogs) in northeastern Canada (Lavoie and Rochefort, 1996; Girard *et al.*, 2002; Lavoie *et al.*, 2003; Poulin *et al.*, 2005), but little research has addressed abandoned peatlands with minerotrophic residual peat. Moreover, the restoration of

fens is still less widespread, as fens are frequently very fertile agricultural land (Höper *et al.*, 2008). The main limitation for bog restoration is early establishment of *Sphagnum* as little natural regeneration of *Sphagnum* mats occurs on peat fields that have been extracted by vacuum methods (Lavoie *et al.*, 2003) because of greater degradation of hydrological conditions from mechanized extraction and complete removal of plant material (Höper *et al.*, 2008). Ferland and Rochefort (1997) found that the vacuumed peat surfaces dry out quickly even if the water table is close to the surface hindering the ability to *Sphagnum* (and other mosses) to re-establish on the peat surface. The presence of vascular plants might be important as it has been found that the growth of *Sphagnum* is strongly dependent on the presence of these plant species at the early stage of restoration (Groeneveld *et al.*, 2007).

According to Graf (2008), when considering ecosystem restoration, there is the spontaneous species pool and the species pool at the natural site. The restored site will have some novel species pool made up of individuals from both these pools and we may choose which species we want in this restored species pool based on the functions of the species (Figure -1.1). My research aims to investigate the C flux functions of the spontaneous species pool to see if there are species that are appropriate to include in the restored pool. Thus, spontaneously recolonizing species could play an important role in the North American restoration process if these species are found to have potential as a C sink, as this is one of the major goals of peatland restoration.

Table – 1.1: Main terms used in the English-language peatland literature (Charman, 2002).

Wetland	<i>Land with the water table close to or above the surface or which is saturated for a significant period of time. Includes most peatlands but also ecosystems on mineral substrates, flowing and shallow waters.</i>
Peatland	<i>Any ecosystem where in excess of 30-40 cm of peat has formed. Includes some wetlands but also organic soils where aquatic processes may not be operating (e.g. drained or afforested peatlands)</i>
Mire	<i>All ecosystems described in English as swamp, bog, fen, moor, muskeg, and peatland (Gore, 1983), but often used synonymously with peatlands (Heathwaite et al., 1993). Includes all peatlands, but some mires may have a mineral substrate.</i>
Fen	<i>A mire which is influenced by water from outside its own limits.</i>
Bog	<i>A mire which receives water solely from rain and/or snow falling on to its surface.</i>
Marsh	<i>Loose term usually referring to a fen with tall herbaceous vegetation often with a mineral substrate.</i>
Swamp	<i>Loose term with very wide range of usage. Usually referring to a fen and often implying forest cover.</i>

Table – 1.2: Important features of the acrotelm (upper layer) and catotelm (lower layer) in peatlands (Based on Ingram, 1978; Ivanov, 1981; Modified from Charman, 2002)

Character	Acrotelm	Catotelm
Water table	Fluctuating	Absent
Water content	Variable	Constant
Aeration	Periodically oxic	Anoxic
Microbial activity	High with oxic and anoxic activity present	Low with only anoxic activity present
Water movement	Relatively fast. Variable from surface to base of acrotelm	Very slow, constant
Exchange of energy and matter	Rapid	Slow
Hydraulic Conductivity	High	Low
Root system	Dense root systems and a diversity of invertebrates	Much reduced vascular rooting, very few invertebrates.

Table – 1.3: Methane emission \pm SE ($\mu\text{mol g}^{-1} \text{day}^{-1}$) for vascular plants found in Buck Hollow Bog and Big Cassandra Bog (May-June, 1992) (Shannon *et al.*, 1996.)

Species	Methane Emission
<i>Chamadaphne calyculata</i>	20 ± 3
<i>Scirpus cyperinus</i>	77 ± 25
<i>Carex oligosperma</i>	91 ± 38
<i>Scheuchzeria palustris</i>	130 ± 14
<i>Calla palustris</i>	280 ± 37

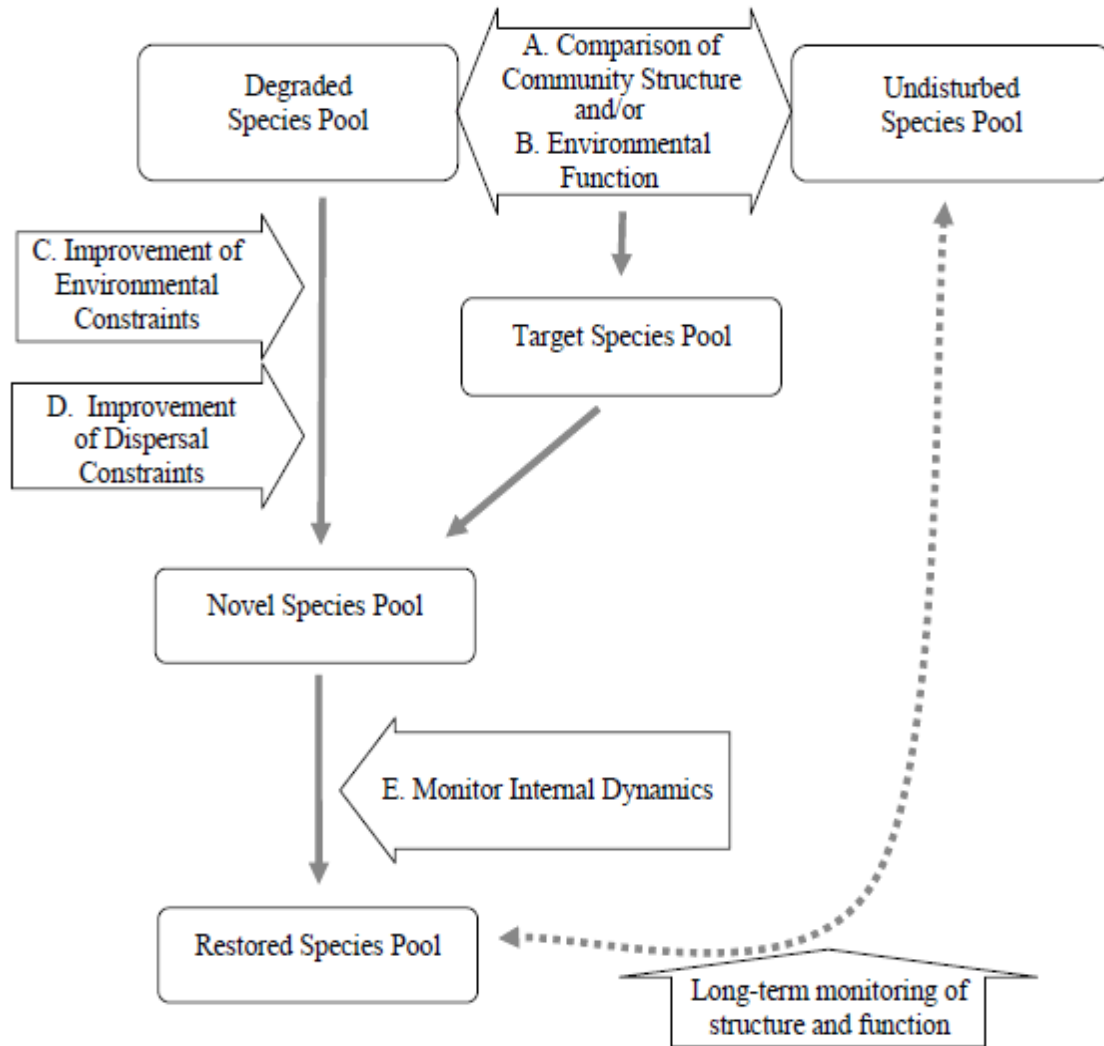


Figure – 1.1: Flow diagram of the processes to identify target species pool for re-vegetation at a moderately-rich fen (Reproduced from Graf, 2008 with permission from the author)

CHAPTER TWO: METHODOLOGY

2.1. The Project and Study Site

This study was part of a large research project where researchers from different field, for example ecology, hydrology, and bio-geosciences worked together to develop a scientifically and industrially accepted fen restoration method. My research focused only on C dynamics of the cutover site, bare peat and natural site. Researchers from University of Laval and University of Waterloo were responsible for ecological association of different plant communities and hydrological restoration for restored site respectively. C dynamics of complex vegetation communities is still under investigation by our research group.

The study was conducted in the Bic Saint-Fabien (BSF) peatland (48°18' N, 68°52' W) which is located approximately 25 km west of Rimouski, Quebec, Canada (Figure -2.1). The undisturbed part of the peatland, a moderately-rich fen with an average peat depth of 4.5 m, is dominated by Eastern White Cedar (*Thuja occidentalis*) in the forested part, with the lower moss layer dominated by brown mosses including *Campyllum stellatum*, *Drepanocladus* spp., and *Tomenthypnum nitens*. In the specific area of the fen investigated, hummocks are dominated by *Sphagnum fuscum* and cover less than 50% of the ground surface. Shrubs are also found on hummocks and include species such as *Chamaedaphne calyculata*, *Kalmia angustifolia*, and *Ledum groenlandicum*. Sedges, such as *Trichophorum cespitosum*, *Trichophorum alpinum*, *Carex interior*, *Carex prairea* and small herbs, including, *Valerianna uliginosa*, *Menyanthes trifoliata*, *Sanguisorba canadensis*, *Parnassia glauca* and *Tofieldia glutinosa*, dominated on the hollows. Some hollow areas also had substantial shrub cover consisting of mainly *Myrica gale*.

The extracted portion of the peatland has been mined for horticultural peat since 1946, with the vacuum extraction technique employed since 1970. Most of the site has been

abandoned during the last few decades and spontaneous recolonization has occurred in some sections. The present harvested portion of BSF is about 22 ha. The south-west one-third of the harvested site has been naturally revegetated by marsh-like vegetation with ruderal species. The remaining two-third has had limited recovery of vegetation, resulting in large regions of bare peats, interspersed with vegetation patches, consisting of a limited number of vascular species. This section will be the focus of large-scale restoration efforts and will be referred to as the cutover site in this study. Overall, on the cutover site, moss cover is very low (<1%). Dominant species in the cutover site are *Scirpus atrocinctus*, *Scirpus cyperinus*, *Equisetum arvense*, *Calamagrostis canadensis*, *Eriophorum vaginatum*, *Salix* spp., *Eupatorium maculatum*, *Spiraea latifolia*, *Hypericum virginicum*, *Carex aquatilis*, and in ditches *Typha latifolia*.

2.2. Carbon Flux

Objective I was to measure the C flux of representative species at an abandoned mined peatland and objective II was to compare the C flux of these species with that of bare areas and natural peatland to determine the effect of mining and revegetation on carbon fluxes. Based on knowledge from previous studies and different literature it was hypothesized for CO₂ that 1) the natural peatland will act as a CO₂ sink whereas bare peat areas of the cutover peatland will be larger sources of CO₂, 2) sites that have been spontaneously recolonized may act as CO₂ sinks during the growing season if biomass is high, and 3) water table as well as soil and air temperature will act as controlling factors for CO₂ uptake. For CH₄, it was hypothesized that 1) the recolonizing vegetation will emit similar or less CH₄ than natural vegetation as cutover site water table is deep, but will have higher fluxes than bare peat as vegetation is supplying fresh substrate to the site, 2) the CH₄ flux will increase over the

growing season because of increasing biomass, and 3) water table and temperature will control the CH₄ emission rate for any given species.

The study was conducted in two growing seasons in 2008 (June-August), and 2009 (May-August). Most of the time CO₂ flux data was collected every week during the study with a few exceptions. CH₄ gas samples were collected 4-8 times for recolonized and natural vegetation but only one time for bare peat in 2008, and 13 times for vegetation collars and 12 times for bare peat in 2009.

Following a preliminary survey of vegetation communities at the cutover peatland at BSF in 2008, five recolonizing species were chosen for study. These were – (1) *Scirpus atrocinctus* (Sci_atr), (2) *Equisetum arvense* (Equ_arv), (3) *Calamagrostis canadensis* (Cal_can), (4) *Eriophorum vaginatum* (Eri_vag), and (5) *Carex aquatilis* (Car_aqu). *Typha latifolia* (Typ_lat) was added in 2009 as it was observed to dominate in ditches which continue to cover a large area at the cutover site. Triplicate plots were established in each vegetation type. Four plots were established on bare peat (P) at the cutover site and studied in both seasons. Six plots were also studied in the natural (N) fen adjacent to the cutover site and arranged to be representative of the hydrologic gradient at the site with three plots on low-lying hollows (Nhol) and three plots at higher, drier hummocks (Nhum). Three more plots were added in 2009 to study the shrub vegetation (Nshrub) common in some hollows at the natural site. See Figure 2.1 for distribution of sampling collars across the study site.

Measurement of C fluxes comprises both CO₂ flux and CH₄ flux. To determine the instantaneous CO₂ exchange rates and collection of CH₄ gas samples the closed chamber method was used (Alm *et al.*, 2007). Briefly, a permanent 60×60 cm square or a round ($d = 30$ cm) collar with a groove for water sealing was installed in soil. The sleeve of the collar extended 20 cm in soil, which is usually below the rooting zone (Laiho and Finér, 1996). For

CO₂ flux measurement a clear acrylic glass chamber (60×60×30 cm) was used. For CH₄ gas sampling an opaque steel chamber of the same size was placed upon the collar creating an airtight seal by adding water in the groove. Ecosystem respiration was measured using the clear chamber by shading it with a black tarp. For measuring the bare peat respiration and CH₄ flux, a round chamber (12.7423 cm³) was used. During the latter part of the season when some vegetation communities (e.g. Typ_lat, Sci_atr) were too high an extension chamber (60×60×60 cm) was added to take the measurement or sample. All the chambers were equipped with a battery-operated fan to mix the chamber headspace air.

2.2.1. CO₂ Flux Measurement

Instantaneous net ecosystem exchange (NEE) is the balance between the simultaneous CO₂ fixation of ground vegetation by photosynthesis and CO₂ release from the system by the respiration of plants and heterotrophes (Alm *et al.*, 2007). For measuring instantaneous CO₂ concentration in the air sealed chamber an infrared gas analyzer (PP systems, EGM-4) was used. The chamber was placed into the groove of the collar and water was added to provide an airtight seal at the beginning of each measurement. The chamber was removed for a while after each measurement to allow gas concentration and temperature to stabilize to the ambient air. The analyzer readings in ppm were monitored and noted every 15 seconds after closing the chamber. The measurement period lasted for 75-105 seconds. Photosynthetic Active Radiation (PAR) was measured with a sensor (PP Systems, USA) located at the top of the chamber and air temperature inside the chamber (°C) with a thermometer (VWR Int., USA) was also recorded at the same time interval during NEE measurements in light. Shades that blocked various amounts of light were also used so that NEE could be modeled throughout the day. Using a spreadsheet I calculated NEE from the linear change of CO₂ concentration in the

chamber headspace in time as a function of chamber volume, air temperature, and air pressure according to the ideal gas law (Alm *et al.*, 2007).

2.2.2. CH₄ Flux Measurement

Measurement of CH₄ flux involves two steps – 1) gas sampling, and 2) gas concentration analysis. For gas sampling an opaque steel chamber was placed upon the collar. Four 20 ml gas samples were collected from the headspace with three-way stopcock syringes at 7, 15, 25, and 35 minutes after sealing the chamber. The samples were then transferred to pre-evacuated Exetainers (Labco Ltd., UK). Samples were sent back to the Department of Geography, University of Calgary for analysis. Air temperature inside the chamber was also measured at the same time the gas samples were collected using a thermometer (VWR Int., USA). Two ambient air samples were also collected for each day of sample collection to use as the CH₄ concentration at the beginning of sample collection (i.e. 0 minute).

The gas samples were analyzed using a Varian Gas Chromatograph 3800 (GC) equipped with a flame ionization detector for CH₄ concentration. To calibrate for any potential instrumental errors, known standards were measured at the beginning and end of an 8 sample analysis pattern. The resultant CH₄ concentrations were then entered into *Microsoft Excel* for further analysis.

2.3. Pore Water CH₄ Measurement

Based on the literature review it was hypothesized that CH₄ flux will be low at the cutover peatland due to the deep water table limiting CH₄ production and allowing high rates of CH₄ oxidation. However, in areas with high biomass, substrate may be available for CH₄ production. We can use pore water CH₄ concentration to see whether CH₄ is produced or not. If CH₄ is produced then the concentration of CH₄ will be high at depth.

Pore water CH₄ samples were collected weekly in 2009 from May to August. Pore water depth profiles were collected from 50 cm, 75 cm and 125 cm depth for each vegetation type and bare peat. All other collars had samplers installed at 50 cm depth. Pore water was collected using samplers (see Strack *et al.*, 2004), consisting of a 20 cm length of 2.5 cm inner diameter (i.d.) PVC pipe slotted at the middle 10 cm, covered in Nitex screening to prevent clogging, and sealed at both ends with stoppers. The stopper at one end contained a central hole through which a sampling tube had been fitted. At the other end of the sampling tube a three-way valve was inserted and sealed with household adhesive. Samplers were inserted vertically to the appropriate depth into the peat in middle of May 2009 with the sampling tube extending from the top end of the sampler to the peatland surface to allow water collection. They were left in place throughout the remainder of the study. The entire sampler was filled with water and the valve closed between sampling to prevent air from travelling to the sampling depth.

To collect a pore water sample, 60 ml of water was removed from the sampler to flush it and then a sample of 5 to 20 ml was collected by connecting a sampling syringe. After that, 20 ml of ambient air was added with the sample and shaken for ~5 minutes to allow equilibration of dissolved gases into the headspace in the syringe. The air sample was then transferred in a pre-evacuated Exetainer (Labco Ltd. UK). Both the water sample collecting time and equilibration time was noted to find the equilibration temperature from meteorological data. Samples were returned to the laboratory and analysed for CH₄ concentration on the Varian gas chromatograph. CH₄ concentration in the pore water was estimated based on the concentration measured in the headspace air sample according to EPA (2001).

2.4. Vegetation Data Collection

“Fuel Rule” appears to provide quick and accurate estimation of vegetation development which can be used to examine C dynamics of the peatland (Davies *et al.*, 2008; Strack and Srivastava, 2010). Thus, I determined volume, a parameter estimated by the *PObscured* program, to make estimation of biomass by using the visual obstruction method developed by Davies *et al.* (2008) known as the ‘Fuel Rule’. Briefly, Fuel Rule is a 2 m stick that is a 2.5 cm wide and painted with alternating red and white bands. One side of the stick has bands of 10 cm width whereas the other side has two bandwidths of 2 and 5 cm starting at opposite ends and running half its length. Each set of bands was labelled in numbers. Depending on the height of the vegetation a scale was chosen for the measurement so that at least five bands were partially obscured.

All vegetation collars were measured using Fuel Rule every two weeks throughout the field season from May to August, 2009. In order to take a reading, the stick was placed vertically into a stand of vegetation within the middle of the 60×60 cm collar. The Fuel Rule was then aligned to the vegetation allowing the vegetation to partially or fully cover at least five bands. Holding the Fuel Rule at arm’s length with the obscured bands at eye level, the percentage obscured by the vegetation was estimated. Visual estimation of the ground surface was conducted by looking downward in the collar and recording the percentage of vascular vegetation and mosses.

The Fuel Rule data was then analyzed in the same method described by Davies *et al.* (2008). A calibration curve relating volume determined by the Fuel Rule to biomass was developed by Strack and Srivastava (2010) for the BSF site. Thus, volume estimates determined for each collar were used as an index for vegetation biomass and used to investigate variations between collars and throughout the growing season.

2.5. Environmental Variables

Soil temperatures (°C) for CO₂ at 2, 5, 10, 15, 20 cm and for CH₄ 2, 5, 10, 15, 20, 25, 30 cm below the surface were measured using a thermocouple thermometer near the base of the closed chamber. Water level relative to soil surface was measured in wells adjacent to each collar. Both water table and soil temperature profile were collected in order to relate the CO₂ and CH₄ fluxes to prevailing environmental conditions. Air temperature and PAR data was measured every minute and averaged every 20 minutes using a automatic data logger (CR 10X, Campbell Scientific Inc., USA) at the field meteorological station (Figure 2.1). Precipitation was also measured using a tipping bucket rain gauge throughout the season. Missing and long-term environmental data were collected from Environment Canada, Rimouski station, which is ~ 25 km east of the site (data available: <http://www.climate.weatheroffice.gc.ca>).

2.6. Data Analysis

2.6.1. Vegetation Model

Vegetation volume was determined as a proxy of biomass in 2009 using ‘Fuel Rule’ (discussed above). To incorporate vegetation into the total C flux model I first modeled the vegetation parameter throughout the season for each individual plot using a Gaussian curve fitting technique. The model I used here is modified from Riutta *et al.* (2007) as follows:

$$V = V_{max} \exp \left[-0.5 \left(\frac{JD - JD_{max}}{b} \right)^2 \right] \text{-----} (2.1)$$

Where, V is the vegetation volume, V_{max} is the maximum V during the season, JD is the Julian day (days of a year numbered from 1 to 365), JD_{max} is the timing of V_{max} and b is the width of the curve.

2.6.2. CO₂ Exchange Model

Seasonal C balance, which is the combination of seasonal CO₂ and CH₄ balance, was used to identify spontaneously recolonizing species with the potential for C accumulation. In order to estimate the net seasonal CO₂ exchange two components, Gross Ecosystem Production (GEP) and Total Ecosystem Respiration (R_{TOT}), were determined. GEP is the difference between NEE and R_{TOT}. The PAR data for 20 minute intervals throughout the season from the automatic logger at the meteorological site and vegetation data from model (2.1) were used to estimate total seasonal GEP for the study period using a maximum rate of gross photosynthetic exchange (GP_{max}) and incident PAR and vegetation relationship. In my study I used the PAR data from 2010 instead of 2009 because of the technical problems that caused the automatic data logger in the field to record wrong values during 2009. Generally, over the season in an average the PAR value does not vary so much from year to year.

GEP as a function of PAR and vegetation was modelled by using the following equation-

$$GEP = \frac{Q \times PAR \times GP_{max}}{(Q \times PAR) + GP_{max}} \times [1 - \exp(-aV)] \text{-----} (2.2)$$

(Modified from Bubier *et al.*, 1998 and Riutta *et al.*, 2007)

Where PAR is the measured photosynthetically active radiation ($\mu\text{mol m}^{-2} \text{s}^{-1}$), GP_{max} is the empirically derived maximum rate of gross photosynthetic exchange of CO₂, Q is the initial slope of the GEP vs. PAR curve, V is the vegetation volume, and a is the initial slope of the saturating V response function (see Waddington and Roulet, 1996; Riutta *et al.*, 2007). As in EGM-4 the PAR is measured in $\mu\text{mol m}^{-2} \text{s}^{-1}$, a linear regression equation was used to convert meteorological station data measured in volts into $\mu\text{mol m}^{-2} \text{s}^{-1}$ (Appendix B: Figure – B.1).

Respiration (R_{TOT}) was modelled separately using a linear relationship with vegetation volume and peat temperature measured at 5 cm depth. The model I used to calculate the respiration over the season is-

$$R_{TOT} = a_1T_5 + b_1V + c \text{ ----- (2.3)}$$

Where, T_5 is the temperature at 5 cm depth ($^{\circ}\text{C}$), a_1 is the slope of temperature parameter, V is the vegetation volume, b_1 is slope of vegetation parameter, and c is a constant.

Continuous peat temperature at 5 cm depth was calculated from air temperature of the automatic data logger using a linear regression with a R^2 value of 0.931 and p-value of <0.005 . Although, the equation was derived from the air temperature logger data of 2009, to calculate the seasonal total respiration using the above model (2.3), I used the logger data from 2010 to be more consistent with PAR data. Finally, NEE was determined as the sum of GEP and R_{TOT} .

This research calculated the seasonal C flux of studied vegetation species and bare peat for only 2009 as the required vegetation and temperature data were collected only in this season. However, this study completed intensive research on measured maximum gross ecosystem production (GEP_{max}), respiration (R_{TOT}), and maximum net ecosystem exchange (NEE_{max}) under full light conditions i.e. $PAR > 1000 \mu \text{ mol m}^{-2} \text{ s}^{-1}$ for both years of the study. Because different plant species saturate at different light levels and GP_{max} from equation 2.2 assumes an infinite upper limit for PAR, this study calculated the measured maximum rate of photosynthesis for each individual vegetation plot for $PAR > 1000 \mu \text{ mol m}^{-2} \text{ s}^{-1}$ so that it could compare the maximum photosynthesis rate more realistically (Bubier *et al.*, 1998). I also looked at which factors controlled the flux under these conditions.

2.6.3. CH₄ Flux

The CH₄ concentrations were entered into *Microsoft Excel* to calculate a linear regression slope from the five sequential sub-samples over the sampling time. The slope, which is the flux rate at the collar area, was extrapolated over time and area into a flux estimating $\mu\text{mol m}^{-2} \text{ day}^{-1}$.

If the initial headspace gas concentration estimated was overly high as compared with the ambient air concentration, the sample was discarded because of probable ebullition due to disturbance in placing the chamber. Overall the CH₄ flux was considered if the slope of the regression is > 0.7 or < 0.3 (indicating low to zero flux). The seasonal CH₄ fluxes were determined by averaging the value throughout the season and multiplying by the total number of days of interest.

2.6.4. Pore Water CH₄ Calculation

Pore water CH₄ concentration in the sub-surface is the sum of concentration of CH₄ in headspace air and in water (EPA, 2001). Concentration in headspace air was calculated by using the following equation:

$$\begin{aligned} CH_4 \text{ in } mg(\text{Headspace}) &= GC \text{ concentration of } CH_4(\mu\text{mol}/\text{mol}) \\ &\times \left[\frac{\text{Molar mass of } CH_4 \left(\frac{g}{\text{mol}}\right) \times \text{Std. Temp. } (^\circ\text{K})}{22.414 \left(\frac{L}{\text{mol}}\right) \times \text{Air Temp. } (^\circ\text{K})} \right] \\ &\times \left[\frac{1000 \left(\frac{\mu\text{mol}}{\text{mol}}\right) \times \frac{g}{mg}}{\quad} \right] \end{aligned}$$

----- (2.5) (EPA, 2001)

Concentration of CH₄ in water was calculated as follows:

$$CH_4 \text{ in mg(water)} = \frac{GC \text{ concentration of } CH_4 \left(\frac{\mu\text{mol}}{\text{mol}} \right) \times 55.5 \text{ mol} \times \text{Molar mass of } CH_4 \left(\frac{\text{g}}{\text{mol}} \right)}{\text{Henry's Constant} \times 1000 \left(\frac{\mu\text{mol}}{\text{mol}} \right) \times \frac{\text{g}}{\text{mg}}}$$

----- (2.6) (EPA, 2001)

Where, 1 L of water = 55.5 moles.

Air temperature during the pore water sample equilibration was extracted from the automatic data logger.

2.6.5. Total Carbon and Greenhouse Gas Balance

Total C balance for the growing season was calculated by adding both total seasonal CO₂ and CH₄ as gram carbon. Annual C balance was calculated by incorporating estimates of missing winter time CO₂ and CH₄ flux and annual DOC/DIC leaching from previous studies. This study also incorporated global warming potential (GWP) for greenhouse gases over a 100 year time frame and calculated the C balance. For example, according to IPCC (2007) the GWP for CH₄ is 25 times CO₂-e over 100 years time period and thus I multiplied the seasonal and/or annual CH₄ flux by 25 and then calculated the total greenhouse gas balance.

2.7. Statistical Analysis

The statistical analysis was performed using *Microsoft Excel*, *Sigmaplot-11.0* and *Minitab 14*. Non-parametric Mood's Median test was applied on the CH₄ flux and pore water CH₄ value or grouped mean fluxes, against vegetation type to test for significant differences as the data were not normally distributed. Treat *et al.* (2007) found the strongest correlation between environmental variables and CH₄ flux using monthly mean values. In this study I used both seasonal and monthly mean value of CH₄ flux and environmental variables.

Multiple regressions were used to see the combined effect of controlling factors such as water table and vegetation on CH₄ flux. The significant difference between the pore water CH₄ concentrations at 50 cm was tested by Mood's median test and one-way ANOVA analysis. Pore water CH₄ concentration at different depths was plotted using a trend line graph to see their concentration trend with depth. To see whether pore water CH₄ concentration has any relation with surface CH₄ flux or vegetation linear regression and Pearson correlation was performed. One-way ANOVA was performed to test the significance of differences between vegetation types for measured CO₂ flux as the data were normal. The relation between vegetation volume, water table, and temperature and measured CO₂ flux was determined by Pearson correlation analysis.

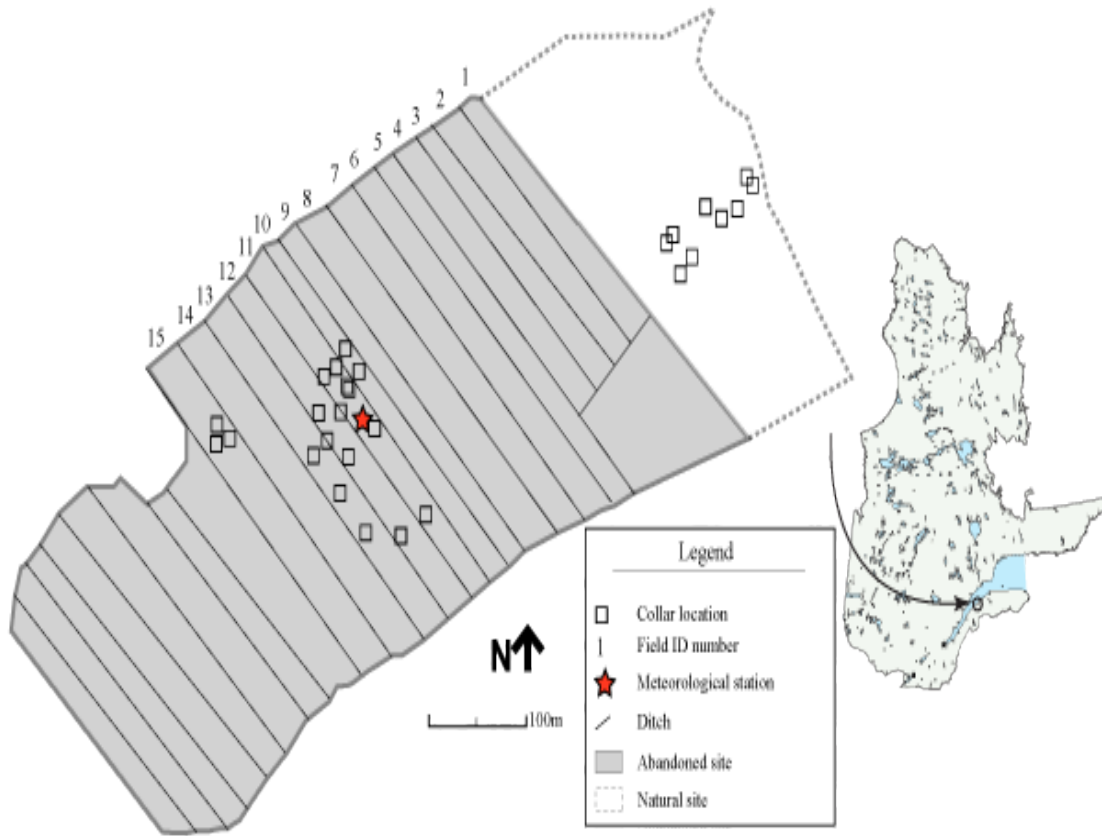


Figure – 2.1: The Bic-St. Fabien peatland study Site. The cutover site is surrounded by natural minerotrophic peatlands to the northeast (beyond the dotted line), but here it only shows the portion of the natural fen where study collars are located.

CHAPTER THREE: RESULTS AND DISCUSSION - METHANE FLUX AND PORE WATER METHANE DYNAMICS

3.1. Results

3.1.1. Environmental Condition

The last 30 years (1981-2010) monthly mean environmental parameters (mean temperature and precipitation) were collected from the Environment Canada Rimouski Station, located approximately 25 km east of the study site. Although the meteorological station is a little far from the site, it is assumed that over the long period (here 30 years) the climatic condition was similar at the study site. However, precipitation data were collected during the study period at the site. This was important as the site is situated in the valley beside a local slope which acts to help create a hotspot for precipitation and thus the precipitation differed from the Environment Canada meteorological station.

The growing season 2008 was slightly warmer (15.65 ± 4.21 °C) than the 30 year average (15.24 ± 3.77 °C), especially during the middle of summer (Table – 3.1). Temperature was colder than the long-term average in 2009 (14.85 ± 3.71 °C) throughout the season except in late summer when temperature was almost the same as the long-term average (Table – 3.1).

The long-term precipitation in the region was almost the same in every month throughout the growing season (Table – 3.1). The long-term (1981-2010) average total precipitation for the growing season (May - August) was 342.87 mm which was much greater than 2008 (268.1 mm). However, 2009 (328.33 mm) was almost same as the long-term average. In monthly average, 2008 was below the long-term average in every month except almost equal in August (Table – 3.1). But in 2009, in July the site received more than two times, in May and August approximately half, and in June almost same amount of precipitation as the long-term average (Table – 3.1).

3.1.2. Plot Characteristics

Vegetation Volume

The highest seasonal mean (\pm standard deviation) vegetation volume was observed for Car_aqu (27.09 ± 11.24) and Sci_atr (27.09 ± 12.58) and lowest in Nhum (4.04 ± 2.77) at the study site (Table-3.1). The seasonal mean vegetation volume varied from 8.44 ± 6.44 (Equ_arv) to $27.09 \pm 11.24/12.58$ (Car_aqu/Sci_atr) for recolonized vegetation. Within natural vegetation types mean values were between 4.04 ± 2.77 for Nhum and 12.41 ± 5.24 for Nshrub.

Water Table

The mean water table varied from -23.04 ± 22.41 cm for Nhum to -6.82 ± 6.69 cm for Nhol in 2008 at the natural site, where negative values indicate the water table below the surface. On the cutover site in 2008 the lowest mean water table was measured at the P sites (-54.18 ± 4.83 cm) and the highest water table was measured at Car_aqu (-0.78 ± 1.77 cm). In 2009 the lowest mean water table of -25.39 ± 16.26 cm was measured at the P sites and the highest mean of 6.9 ± 2.85 cm at Car_aqu plots on the cutover site. Natural site water table in 2009 was -29.20 ± 10.68 cm, -6.91 ± 3.99 cm and -6.68 ± 4.37 at Nhum, Nhol and Nshrub respectively (Table-3.2).

Soil Temperature

The mean soil temperature did not differ much between vegetation types in either year of the study. The highest mean temperature at 5 cm depth was 19.8 ± 2.27 °C at Cal_can and the lowest was 17.7 ± 1.69 °C at Sci_atr and at 30 cm depth temperature ranged from 15.1 ± 0.26 °C at P to 16.8 ± 2.37 °C at Cal_can in 2008. Mean temperatures were slightly cooler in 2009 varying between 19.7 ± 4.06 °C at P and 16.8 ± 3.20 °C at Car_aqu at 5 cm depth; and 15.8 ± 4.00 °C at Nhol and 12.8 ± 5.15 °C at Sci_atr at 30 cm depth (Table – 3.2).

3.1.3. CH₄ Flux

The CH₄ flux varied between vegetation types in both years of study (Table-3.3). Seasonal mean (\pm standard deviation) CH₄ flux ranged from $7.72 \pm 10.31 \mu\text{mol m}^{-2} \text{day}^{-1}$ to $7423.83 \pm 2226 \mu\text{mol m}^{-2} \text{day}^{-1}$ and $6.80 \pm 20.30 \mu\text{mol m}^{-2} \text{day}^{-1}$ to $14497.40 \pm 6143 \mu\text{mol m}^{-2} \text{day}^{-1}$ in 2008 and 2009, respectively.

Among the recolonizing and natural fen communities the median flux was significantly higher at Car_aqu and Eri_vag than all other vegetation types in 2008 (Figure-3.1; Mood's median, Chi-square = 33.51; DF = 7; $p = 0.00$). There were no statistically significant differences among the remaining recolonizing and natural plot types. In 2009 the CH₄ flux for Typ_lat was significantly higher compared to Car_aqu and Eri_vag although their flux was still higher than other species (Figure-3.1; Mood's median, Chi-Square = 130.52; DF = 9; $p = 0.00$). It is important to note that in 2009 the flux of Car_aqu was almost half that of the previous year despite the fact that other species showed a slight rise in flux.

The temporal variation of CH₄ flux in 2008 showed that the CH₄ flux increased throughout the growing season until July and then declined in August for most of the species except for Equ_arv and Nhum (Figure-3.2a, b). Hummocks (Nhum) showed a reverse pattern with a decline in flux during the growing season and increase at the end of the season, while Equ_arv had a continuous decline. Given the relatively small CH₄ fluxes at both these vegetation communities, there was actually little change in flux over the season. In 2009, a similar pattern appeared with increases throughout the growing season until July, descending in August (Figure-3.2c, d) except for Typ_lat, where fluxes continuously increased.

3.1.4. Pore Water CH₄

Pore water samples were only collected in 2009. The pore water CH₄ concentration at 50 cm varied significantly between plot types (Figure-3.3; Mood's median, Chi-square = 69.0; DF = 8; p = 0.00) with the highest values at P and lowest at Sci_atr. Among the vegetated plots the mean value ranged from 23.85 ± 23.16 μM for Sci_atr to 569.25 ± 364.61 μM at Nhol. Natural vegetation communities had higher pore water CH₄ concentrations at 50 cm depth than all recolonizing vegetation types at the cutover site except for Eri_vag which was similar to Nhum (Figure-3.4). Other than significantly higher concentration at Eri_vag, all recolonizing vegetation at the cutover site had similar pore water concentration. There was no significant correlation between pore water CH₄ concentration at 50 cm depth and CH₄ flux (Pearson correlation -0.265, p = 0.221).

Both increasing and decreasing trends were found in depth profiles of pore water CH₄ between 50 and 125 cm below the surface (Figure-3.4). The concentration of pore water CH₄ increased with depth for Sci_atr, Typ_lat and Nhum. For P the concentration decreased sharply with depth. Both Cal_can and Nhol had a sharp decrease from 50 to 75 cm, below which there was almost no change with depth.

3.1.5. Potential Controls on CH₄ Flux

When I included values from all individual flux measurements I found significant correlations (Pearson correlation, p < 0.05) between water table and CH₄ flux at Cal_can in 2008 and Typ_lat, Nhol, and P in 2009 where the latter was negatively correlated (Table-3.4). Treat *et al.* (2007) found stronger correlations between CH₄ flux and environmental variables using monthly means, compared to daily flux measurements, so I also chose to investigate further correlations using monthly mean values. In my study I found significant correlation between monthly mean water table and CH₄ flux for only Car_aqu (Pearson correlation 0.602,

$p < 0.05$) and Nhol (Pearson Correlation 0.609, $p < 0.05$) in 2009. The linear regression between monthly mean water table and CH₄ flux also found the water table effect on Car_aqu ($R^2 = 0.362$; $p < 0.05$), and Nhol ($R^2 = 0.370$; $p < 0.05$) in 2009 (Table-3.5). However, I did not find any water table effect for individual vegetation types in 2008. I found a significant relationship between seasonal mean CH₄ flux and water table across the entire site using the mean value of each individual plot in 2008 ($R^2 = 0.322$; $p < 0.05$) and 2009 ($R^2 = 0.265$; $p < 0.05$). When I excluded Typ_lat in my 2009 analysis, water table explained slightly more of the variability ($R^2 = 0.344$; $p < 0.05$) (Figure -3.5a).

In 2008, soil temperature was only correlated to CH₄ flux at Eri_vag where all depths from 10 cm to 30 cm had a significant correlation ($p < 0.05$). In 2009, Typ_lat, Sci_atr, and Eri_vag showed significant correlation ($p < 0.05$) between soil temperature at every depth (2 – 30 cm) and CH₄ flux. However, for Car_aqu I found this correlation only from 10 cm to 30 cm depth for 2009. Overall, correlations between soil temperature and CH₄ flux became stronger with depth of temperature measurement. No significant correlations were found between air temperature and CH₄ flux in 2008, but Eri_vag and all natural vegetation (i.e. Nhum, Nhol, and Nshrub) showed significant correlation ($p < 0.05$) in 2009.

For the whole site I found a strong correlation between CH₄ flux and vegetation volume across at vegetated collars (Pearson correlation 0.706, $p < 0.01$). This relation is also confirmed by the linear regression of monthly mean vegetation volume and CH₄ flux ($R^2 = 0.498$; $p < 0.01$). When I tested this for individual vegetation types I found significant regressions between vegetation volume and CH₄ flux for Car_aqu, Typ_lat, Eri_vag, and Nhol (Table-3.5). I also found significant relationship across the study site using seasonal mean CH₄ flux and vegetation volume ($R^2 = 0.232$; $p < 0.05$). Vegetation volume explained more of the variability in CH₄ flux between plots ($R^2 = 0.363$; $p < 0.01$) when I excluded Typ_lat from

the analysis (Figure-3.5b). No significant relationship was observed between the mean vegetation volume and pore water CH₄ concentration at 50 cm depth.

A significant multiple regression using monthly mean with water table and vegetation volume effects on CH₄ flux was observed for Car_aqu and Typ_lat; however, the main control on CH₄ flux in these cases appears to be vegetation as the inclusion of water table explains little more variability in CH₄ flux than volume alone (Table-3.5). In contrast, a significant multiple regressions was found for Nhol explaining 67.7% of variability in CH₄ flux, while water table and vegetation volume alone explained only 37 and 40.1%, respectively (Table-3.5). However, when I ran flux, water table and vegetation volume of individual plots, I found significant multiple regression across the entire study site ($R^2 = 0.338$; $p < 0.01$). The effect appeared stronger ($R^2 = 0.517$; $p < 0.01$) when I excluded Typ_lat from the analysis.

3.2. Discussion

3.2.1. CH₄ flux and its Controlling Factors

In peatlands, CH₄ fluxes vary from slight uptake to efflux of more than 65,000 $\mu\text{mol m}^{-2} \text{day}^{-1}$ (Klinger *et al.*, 1994). However, fen vegetation types are generally stronger emitters than bogs because the anoxic zone is on average closer to the peatland surface (Moore *et al.*, 1990). CH₄ flux from natural minerotrophic peatlands ranges from 6125 $\mu\text{mol m}^{-2} \text{day}^{-1}$ to 15 562.5 $\mu\text{mol m}^{-2} \text{day}^{-1}$ (Saarnio *et al.*, 2007). The CH₄ fluxes for most of the cutover and natural vegetation communities at the BSF site were substantially lower than the above lowest value except Car_aqu ($7423.83 \pm 2226.88 \mu\text{mol m}^{-2} \text{day}^{-1}$) in 2008 and Typ_lat ($14497.40 \pm 6143 \mu\text{mol m}^{-2} \text{day}^{-1}$) in 2009. Data for bare peat CH₄ flux agreed well with that of Waddington and Day (2007) who reported values from bare peat of a cutover site of $-87.5 \mu\text{mol m}^{-2} \text{day}^{-1}$ to $6.25 \mu\text{mol m}^{-2} \text{day}^{-1}$.

In general, extraction of peatlands decreases CH₄ flux due to drainage, except at ditches where fluxes can remain high (Sundh *et al.*, 2000). It was hypothesized that the recolonized vegetation will emit similar or less CH₄ than natural vegetation but will have higher fluxes than bare peat as vegetation is supplying fresh substrate to the site. This was observed at the BSF peatland with CH₄ flux from bare cutover peat being lowest of all cover types tested, while also having the deepest water table. In this study CH₄ flux from cutover recolonizing vegetation was higher than cutover bare peat and thus the effect of revegetation was increased CH₄ flux. While shallower water table at some revegetated plots may partially explain the increase in CH₄ flux, results of multiple regressions suggest that the increase in biomass also results in higher CH₄ emission. Although one restoration goal is to reduce C and greenhouse gas flux to the atmosphere, the main goal is to return the ecosystem function and thus the system will emit some CH₄ because of its wetland characteristics. On average, measured fluxes were still much lower than the average CH₄ emission rate of minerotrophic peatlands with most vegetation communities on the cutover site having lower efflux than the natural peatland Nhol and Nshrub vegetation types. Similar results were observed in Europe where revegetated cutover peatland CH₄ emissions were lower than adjacent pristine mires (Komulainen *et al.*, 1998; Tuittila *et al.*, 2000a).

In addition to variations in CH₄ flux between vegetation types, there was also noticeable variability between years and over the season. It was apparent that variability in CH₄ flux results from the combined effect of physical (e.g. water table and temperature) and biological factors (e.g. vegetation and microbial communities). As CH₄ is produced only under highly reduced conditions and can be oxidized when oxygen is available, water table position is generally strongly correlated to CH₄ flux in peatlands (Granberg *et al.*, 1997; Bellisario *et al.*, 1999). This study found a similar result for the overall site (Figure -3.6a).

However, within a vegetation type, water table was only significantly related to CH₄ flux for a limited number of vegetation types (Cal_can, Typ_lat, Nhol). This is likely due to the relatively dry nature of the cutover site and the large difference in water table between vegetation types. When water tables are deep, fluxes tend to be very low (Roulet *et al.*, 1993; Wilson *et al.*, 2008) because most CH₄ produced is oxidized as it diffuses through the unsaturated peat. Thus, at BSF, for vegetation types with deep water tables flux was always low, while at the few wet sites (Car_aqu, Typ_lat, Nhol) CH₄ flux was high and varied temporally due to additional factors such as temperature and vegetation biomass. The high CH₄ flux from Typ_lat relative to its water table and vegetation volume likely indicates the importance of ebullition. Thus exclusion of this species in the regression analysis increased the amount of variability explained by water table and vegetation volume both of which are more likely to be important when diffusion dominates. More research on the importance of ebullition in cutover and restored peatlands is required to better characterize CH₄ flux from these sites.

A correlation between soil temperature and CH₄ flux has been observed when water table level is at or close to the surface (Tuittila *et al.*, 2000c; Ding *et al.*, 2004). The findings for Car_aqu, Typ_lat and Nhol are also consistent with this as temperature controls microbial activity and high water table conditions allow for CH₄ production with little oxidation. Soil temperature also played a role for differences in CH₄ flux among vegetation communities at BSF. Sites dominated by vascular, aerenchymatous plants have a stronger positive relationship between CH₄ fluxes and peat temperature at depth than other sites (Saarnio *et al.*, 1998; Shannon and White, 1994). This research also found soil temperature - CH₄ flux relationship for aerenchymatous vegetation types, Sci_atr (*Scirpus atrocinctus*) and Eri_vag (*Eriophorum*

vaginatum) even when the water table was deep. This suggests that deep root systems are venting CH₄ produced below the water table to the atmosphere.

It was observed that presence and productivity of vegetation is important for CH₄ release given the significant relationship between vegetation volume and CH₄ flux found across the site (Figure-3.5b). While vegetation appears to play a role in increasing CH₄ emissions across the entire peatland since fluxes from vegetated collars are higher than bare peat, strong correlations were found only at Car_aqu, Typ_lat, Nhol and Eri_vag (Table-3.5). Several studies have found that vegetation plays a more important role in CH₄ efflux when the water table is close to the surface (Waddington *et al.*, 1996; Strack *et al.*, 2006). Similar results were observed at wet vegetation types Car_aqu, Typ_lat and Nhol, where the shallow water table provides necessary conditions for methanogenesis and vascular vegetation is likely important for provision of fresh substrate. At these sites plant litter deposited at the surface is quickly submerged providing fresh substrate for CH₄ production (Ding *et al.*, 2002). Tuittila *et al.* (2000b) suggested that substrate availability is an important factor in controlling dynamics of CH₄ fluxes at a cutover peatland. Moreover, when the water table is near the surface, a higher proportion of the rooting zone is anoxic, potentially resulting in greater venting of CH₄ to the atmosphere (Waddington *et al.*, 1996).

Substrate availability is likely also responsible for temporal variation of CH₄ flux during growing season. According to Bergman *et al.* (2000) the CH₄ produced in peat soils can vary over the growing season due to variations in the supply of available substrate, the activity of the microbial community or changes in temperature. This study found that CH₄ flux starts to rise when vegetation volume and temperature was high during the peak season and began to decrease when the plants began senescence and temperature decreased later of the season. So, as has been suggested elsewhere, it is probable that the methanogenic

population, in response to substrate supply and temperature, contributed to the observed increase in flux through greater CH₄ production (e.g. Kettunen *et al.*, 1999). The exceptional behaviour of Equ_arv plots in 2008 (continual decline in flux over the season) is mainly controlled by water table position. Though there is substrate available for CH₄ production most would be oxidized due to the deep water tables and lack of plant mediated transport. The continuous rise of CH₄ flux for Typ_lat in 2009 may be because the water table was always above the surface and thus little oxidation would take place while substrate supplied by the vegetation community by litter fall during senescence would be available at this saturated surface.

High flux from Eri_vag plots clearly illustrates the important role that some vascular species can play in CH₄ transport. Despite a deep water table, CH₄ flux from Eri_vag is relatively high and significant correlations between both soil temperature and vegetation volume with CH₄ were observed. Frenzel and Karofeld (2000) suggest that *Eriophorum vaginatum* (Eri_vag) is a deep rooting vascular plant species and responsible for venting CH₄ from the anoxic peat layer, particularly in cutover peatlands (e.g. Marinier *et al.*, 2004). Tuittila *et al.* (2000b) found that increasing primary production and subsequent deposition of substrate to anoxic conditions by this species increased CH₄ flux in a restored cutover site in Europe. A similar result was observed by Waddington and Day (2007) in a Canadian peatland. Plant-mediated transport is likely also important for Car_aqu and Typ_lat as both species have been observed to vent CH₄ to the atmosphere (Joabsson *et al.*, 1999b; Ding, 2002; Laanbroek, 2009; Lai, 2009). However, as water table is close to the surface at these locations, oxidation in the peat is limited suggesting that the vegetation's role in substrate supply may be more important than transport for maintaining high CH₄ flux at these locations. Sci_atr and Cal_can may also provide plant-mediated transport, however, the low fluxes from these locations

suggest that either their ability is limited, or rooting depth is not sufficient to tap anoxic zones where CH₄ is produced. Thus the hypothesis that CH₄ flux will increase over the growing season, and water table and temperature will control the CH₄ emission was supported by these findings.

3.2.2. Pore Water CH₄ and its Controlling Factors.

It was hypothesized that the pore water CH₄ concentration would be determined mostly by vegetation characteristics, such as biomass production and root system, and as well by as environmental factors such as water table, temperature, and atmospheric pressure. The mean pore water CH₄ concentration at BSF site up to 731.22 μM at 50 cm depth is similar to other findings (Clymo and Pearce, 1995; Waddington and Roulet, 1997; Blodau *et al.*, 2007; Strack and Waddington, 2008). Given that moderate concentrations of CH₄ were found across the cutover site, it is clear that substrate exists for methanogenesis and that the low fluxes observed likely result from high levels of oxidation due to deep water table. Surprisingly, the highest mean concentration at the study site at 50 cm depth was found at the bare peat (P) plots. It would be expected that the deep old peat exposed by extraction would have very little substrate remaining to produce CH₄. As this sampling site is near the margin of the cutover peatland it is possible that some substrate was supplied to this site from the surrounding unharvested peatland. Moreover, extraction caused the compaction of the peat, possibly limiting the release of CH₄ once it is produced leading to the observed high concentrations. In contrast, fresh litter and root systems at vegetated plots should increase pore size, encouraging CH₄ release at these locations. Among the natural vegetation communities Nhol had higher concentration than Nhum at 50 cm depth but similar values at 75 cm depth and increasing deeper. This can be explained by the control of water table on pore water CH₄ concentration. CH₄ produced at 50 cm at Nhum is more likely to be oxidized than at Nhol given the deeper

water table position at the former. The pore water concentration of recolonizing vegetation communities at the cutover site is much lower than at the natural fen. The concentration of CH₄ increased with depth for Eri_vag and Sci_atr plots but decreased for Cal_can and Typ_lat although the variation was very small. Popp *et al.* (1999) suggested that the presence of vascular vegetation may play a role in this small-scale variability as roots provide substrate via root litter and exudates and also transport oxygen below the water table. Eri_vag has the deep root system and it supplies more substrate and oxygen to the deep layer where it is fully saturated. Thus these plots have very high CH₄ flux as well as increased pore water CH₄ concentration with depth. Both Typ_lat and Cal_can have similar pore water CH₄ concentration at 50 cm depth. The water table of Typ_lat was at or above the surface and vegetation volume was high which explained their high surface flux and decreasing trends in pore water concentration with depth. On the other hand Cal_can had deeper water table thus most of the CH₄ produced could become oxidized and the substrate could not reach in deeper layers. Thus, this study found low surface flux and sharp decrease of pore water concentration from 50 to 75 cm and then no change at all. However, more research is needed to know which combined factors are actually controlling pore water CH₄ concentration with depth.

Table – 3.1: The long-term (1981-2010), 2008, and 2009 mean temperature and precipitation at BSF. All data was retrieved from Environment Canada, Rimouski, Quebec station, located 25 km east of the site except precipitation data measured for 2008 and 2009 at the meteorological station installed at the site.

Month	Temperature (°C)			Precipitation (mm)		
	<i>1981-2010</i>	<i>2008</i>	<i>2009</i>	<i>1981-2010</i>	<i>2008</i>	<i>2009</i>
May	9.88	10.20	9.60	83.67	61.4	25.73
June	15.40	14.80	14.90	85.83	72.50	79.61
July	18.34	20.00	17.20	88.27	48.40	184.14
August	17.32	17.60	17.70	85.10	85.80	38.85

Table – 3.2: Mean water table, soil temperature (5 and 30 cm depth), and vegetation volume of study plots in 2008 and 2009. Plus minus standard deviations are showing within brackets.

Plot type	Water Table (cm)		Soil Temp. - 5 cm (°C)		Soil Temp. - 30 cm (°C)		Vegetation Volume
	Mean (±SD)		Mean (±SD)		Mean (±SD)		Mean (±SD)
	2008	2009	2008	2009	2008	2009	2009
Car_aqu	-0.8 (1.77)	6.9 (2.85)	17.8 (1.5)	16.8 (3.20)	15.7 (2.5)	14.0 (2.94)	27.09 (11.24)
Cal_can	-31.9 (21.00)	-21.1 (14.01)	19.8 (2.27)	17.9 (4.08)	16.8 (2.37)	15.0 (3.47)	11.36 (7.60)
Equ_arv	-23.1 (10.35)	-18.6 (11.30)	19.6 (2.49)	17.9 (4.52)	16.2 (2.82)	13.6 (5.01)	8.44 (6.44)
Typ_lat	n.m. ^a	1.3 (8.41)	n.m. ^a	17.6 (4.39)	n.m. ^a	14.0 (3.39)	25.88 (11.91)
Sci_atr	-32.5 (20.69)	-17.9 (15.45)	17.7 (1.69)	17.7 (3.73)	15.5 (1.86)	13.9 (3.44)	27.09 (12.58)
Eri_vag	-28.7 (13.94)	-20.4 (12.64)	19.6 (1.82)	16.2 (4.36)	15.7 (2.73)	12.8 (5.15)	17.65 (4.44)
P	-54.2 (4.83)	-25.4 (16.26)	18.6 (0.95)	19.7 (4.06)	15.1 (0.26)	14.5 (4.08)	0
Nhum	-23.0 (22.42)	-29.2 (10.68)	18.2 (2.00)	18.6 (4.91)	15.5 (2.58)	15.8 (4.10)	4.04 (2.77)
Nhol	-6.8(6.69)	-6.9 (3.99)	18.7 (2.00)	19.3 (4.78)	16.1 (2.90)	15.8 (4.00)	6.95 (4.47)
Nshrub	n.m. ^a	-6.7 (4.37)	n.m. ^a	18.9 (5.32)	n.m. ^a	15.7 (4.13)	12.41 (5.24)

a. n.m. – not measured

Table – 3.3: Mean, median and standard deviation (SD) for CH₄ flux in 2008 and 2009 for recolonized, natural vegetation and bare peat plots in $\mu\text{mol m}^{-2} \text{day}^{-1}$.

Plot type	2008			2009		
	Mean	Median	SD	Mean	Median	SD
Car_aqu	7423.8	6428.0	2226.9	3507.4	3384.0	874.1
Cal_can	383.9	303.3	106.3	621.8	551.0	369.6
Equ_arv	238.0	218.0	120.4	897.0	470.0	752.1
Typ_lat	n.m. ^a	n.m. ^a	n.m. ^a	14497.4	9461.0	6143.6
Sci_atr	359.2	188.0	194.4	1014.2	659.0	116.0
Eri_vag	2202.6	2403.0	270.8	2051.9	1786.0	356.4
P	7.72	8.58	10.3	6.80	1.11	20.3
Nhum	132.7	68.7	76.9	203.1	114.9	109.5
Nhol	478.8	385.0	70.8	978.9	726.0	291.5
Nshrub	n.m. ^a	n.m. ^a	n.m. ^a	425.0	339.5	318.7

a. n.m. – not measured

Table – 3.4: Pearson correlation between mean monthly water table and CH₄ flux in 2008 and 2009.

Plot type	Water table 08	P Value	Water table 09	P value
Car_aqu	-0.154	0.692	0.269	0.108
Cal_can	0.462	0.053	0.223	0.178
Equ_arv	0.351	0.140	0.247	0.159
Typ_lat	n.m. ^a	n.m. ^a	0.377	0.037
Sci_atr	-0.360	0.187	0.160	0.329
Eri_vag	0.291	0.258	0.079	0.633
P	n.d. ^b	n.d. ^b	-0.298	0.040
Nhum	0.143	0.526	-0.201	0.209
Nhol	0.300	0.186	0.313	0.046
Nshrub	n.m. ^a	n.m. ^a	0.247	0.119

a. n.m. – not measured

b. n.d. – not determined

Table – 3.5: Single and multiple regressions between monthly mean of CH₄ flux, water table and vegetation volume in 2009.

Plot type	Water Table			Volume			Water Table and Vegetation		
	R ² (%)	F	P	R ² (%)	F	P	R ² (%)	F	P
Car_aqu	36.2	5.67	0.039	67.4	20.71	0.001	67.8	9.46	0.006
Cal_can	17.1	2.06	0.182	0.00	0.00	0.992	17.1	0.93	0.430
Equ_arv	10.5	1.17	0.305	30.8	4.44	0.061	34.9	2.41	0.145
Typ_lat	6.3	0.67	0.431	48.3	9.33	0.012	48.4	4.23	0.051
Sci_atr	8.5	0.93	0.357	20.9	2.64	0.135	30.7	1.99	0.192
Eri_vag	5.1	0.54	0.479	32.6	4.84	0.052	39.2	2.91	0.106
Nhum	17.0	2.04	0.183	0.1	0.01	0.917	23.9	1.41	0.292
Nhol	37.0	5.88	0.036	40.1	6.68	0.027	67.7	9.41	0.006
Nshrub	13.0	1.49	0.250	0.50	0.05	0.826	16.4	0.89	0.446

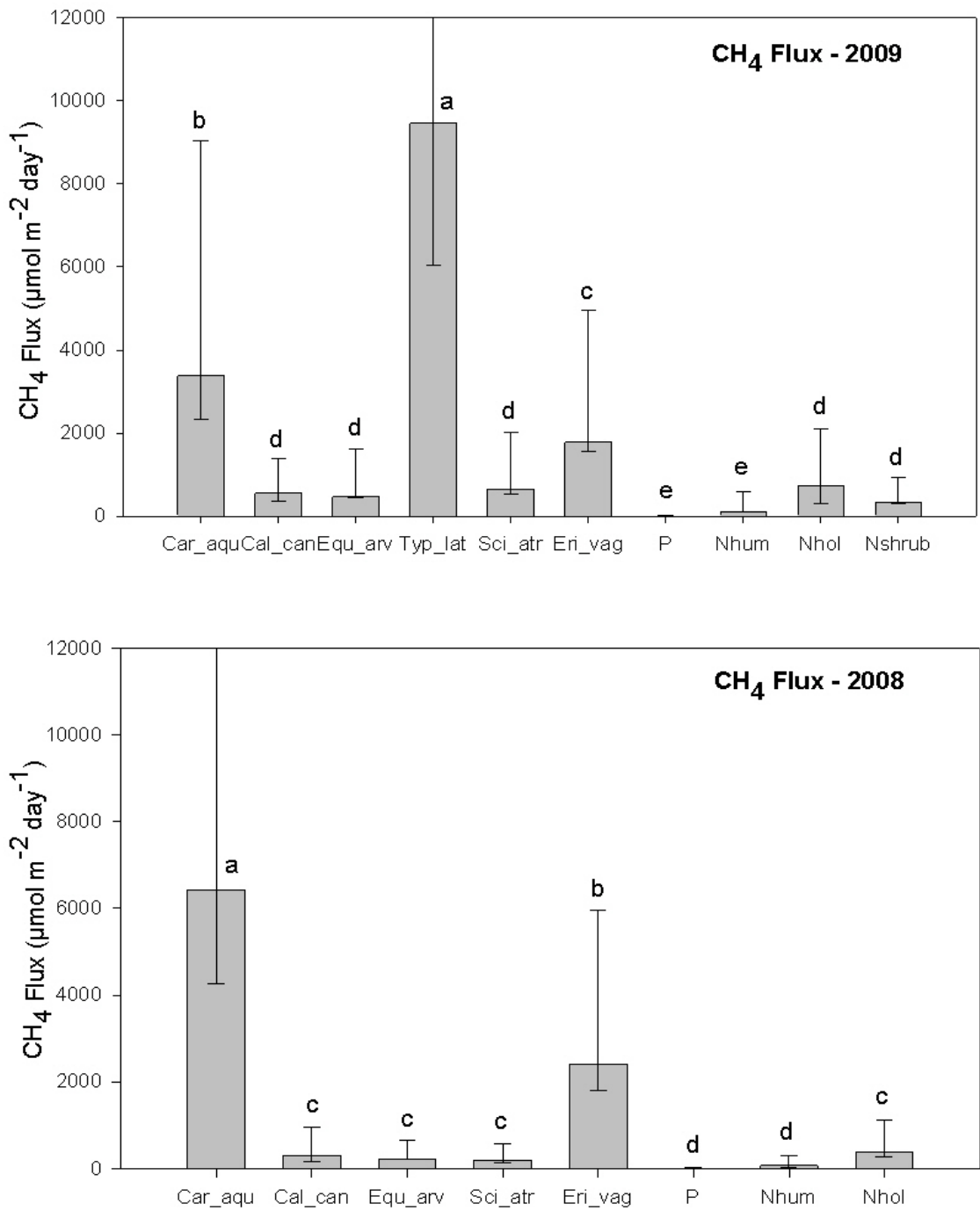


Figure – 3.1: Median growing season CH₄ flux in 2009 and 2008. Error bars give inter-quartile range. Plot types are significantly different from each other if no letters are in common.

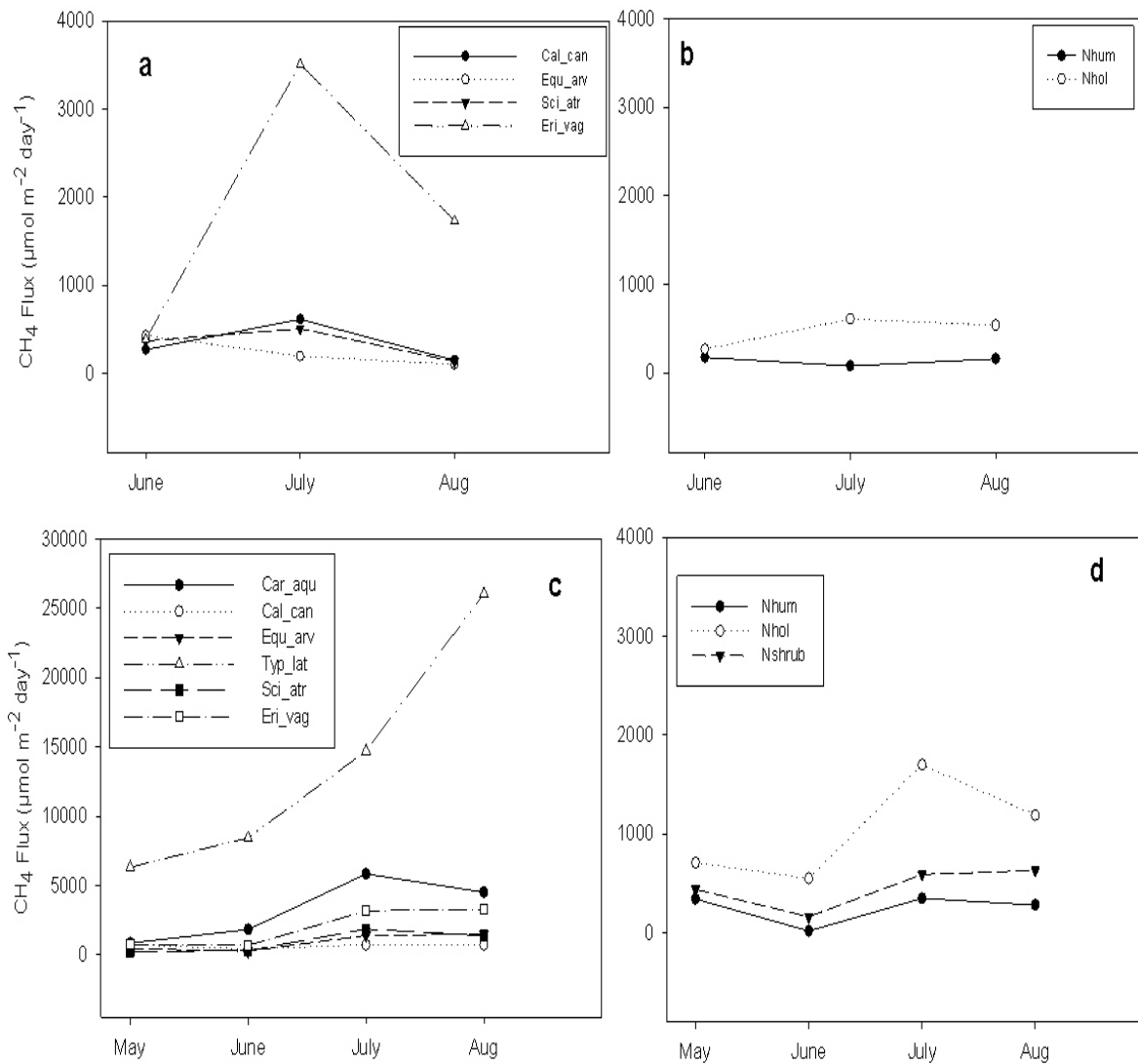


Figure – 3.2: Monthly mean CH₄ flux for **a)** recolonizing communities in 2008, **b)** natural communities in 2008, **c)** recolonizing communities in 2009, and **d)** natural communities in 2009. Error bars are omitted for clarity, but standard deviations of monthly means are similar to the whole growing season deviations given in Table-3.3. Note the different scale on plot c due to high fluxes at Typ_lat.

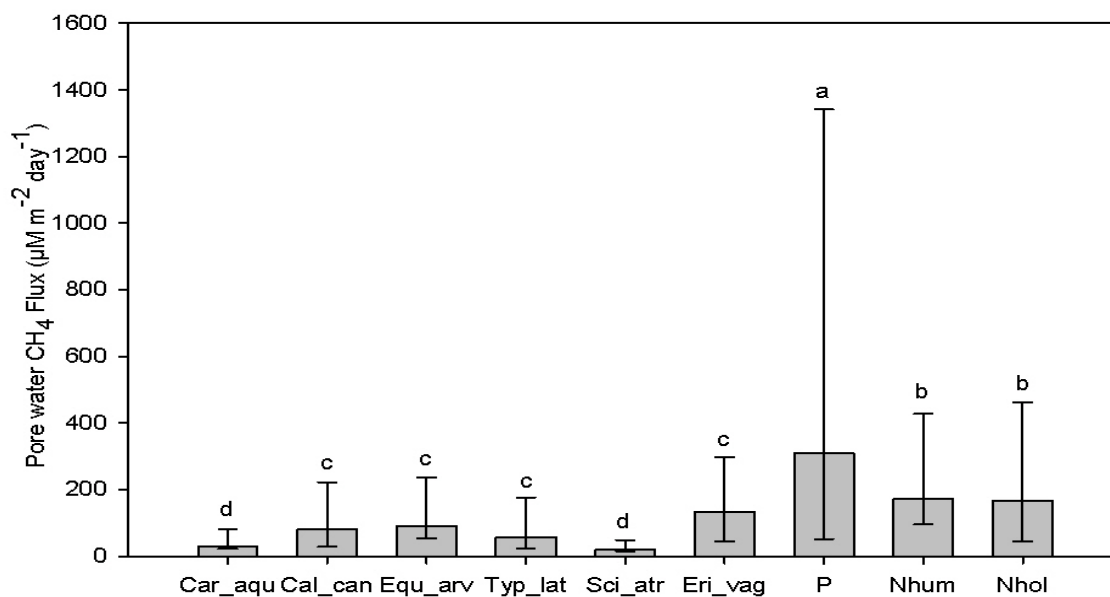


Figure – 3.3: Median pore water CH₄ concentration at 50 cm depth in 2009. Error bars give inter quartile range. Plot types are significantly different from each other if no letters are in common.

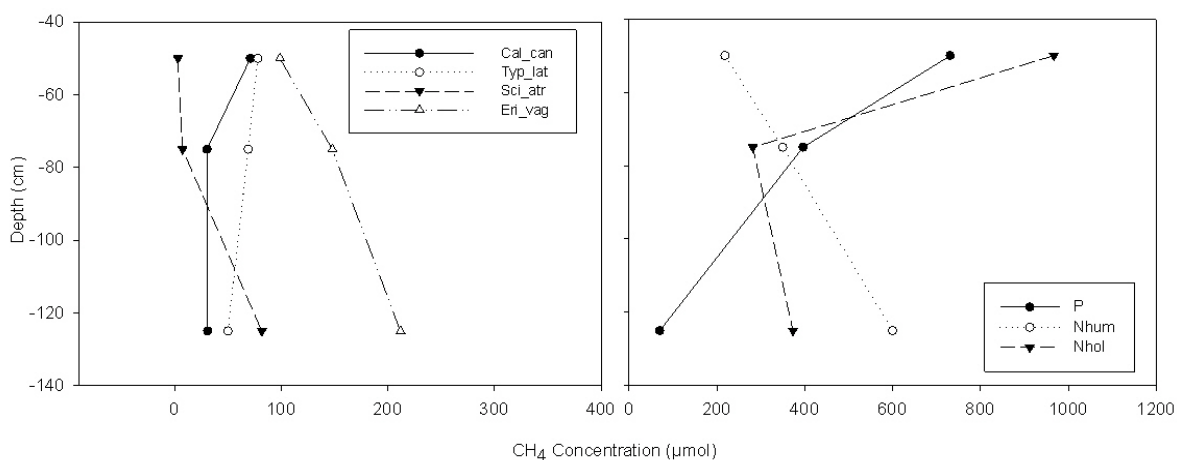


Figure – 3.4: Depth profile of pore water CH₄ concentration at recolonizing communities, bare peat and natural vegetation in 2009. Error bars are omitted for clarity, but standard deviations are similar to those in Figure-3.3.

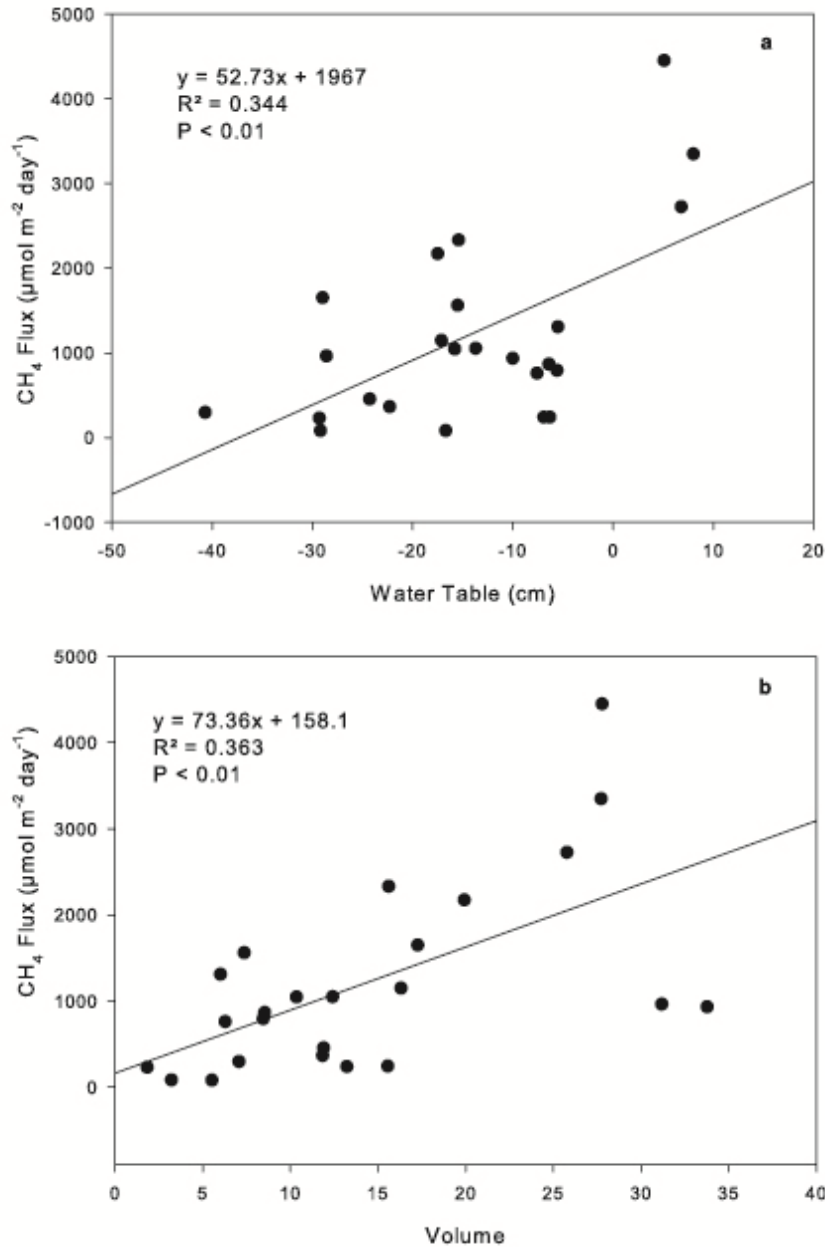


Figure – 3.5: Regression between monthly mean CH₄ flux and vegetation volume in 2009 excluding the value of Typ_lat. Typ_lat has strong correlation with vegetation volume itself but we exclude here because Typ_lat has much higher CH₄ flux compared to other communities, but similar vegetation volume. The inclusion of Typ_lat in the regression gives the equation CH₄ flux = 265.1*volume – 1338, R² = 0.312, p <0.001.

CHAPTER FOUR: RESULTS AND DISCUSSION –

CO₂ DYNAMICS AND CARBON BALANCE

4.1. Measured CO₂ Flux

The study measured instantaneous NEE using a clear glass chamber and in four different light conditions. These are – (a) Full light condition i.e. no shade, (b) 50% blocking the sunlight using one shade (c) 75% blocking the sunlight using two shades, and (d) full dark condition using a black tarp. These different light conditions and their respective NEE were used to model the CO₂ exchange over the season.

4.1.1. Maximum Gross Ecosystem Production (GEP_{max})

The mean GEP_{max} ($PAR \geq 1000 \mu\text{mol m}^{-2} \text{s}^{-1}$) value for revegetated areas of the cutover site was -26.382 ± 2.904 and $-33.013 \pm 6.062 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in 2008 and 2009 respectively. The value for natural site was $-8.935 \pm 1.605 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in 2008 and $-28.827 \pm 2.13 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in 2009. The measured GEP_{max} varied significantly between vegetation types in 2008 (ANOVA – $F = 41.64$, $P < 0.05$; Figure – 4.1) as well as in 2009 (ANOVA – $F = 6.28$, $P < 0.05$; Figure – 4.1). In 2008 the mean GEP_{max} varied from $-7.14 \pm 1.86 \text{ (g CO}_2 \text{ m}^{-2} \text{ day}^{-1})$ at Nhum to $-39.12 \pm 6.24 \text{ (g CO}_2 \text{ m}^{-2} \text{ day}^{-1})$ at Sci_atr (Appendix A: Table – A.1). In 2009, the lowest GEP_{max} value was also at Nhum ($-15.48 \pm 1.79 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) which was two times higher than 2008. The highest GEP_{max} was again at Sci_atr ($-42.04 \pm 6.64 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) in 2009 but the value was similar to the preceding year. Cal_can and Equ_arv had about two times and Nhol had four times higher GEP_{max} in 2009 than 2008 while the other vegetation types had similar GEP_{max} in both years (Appendix A: Table – A.1). All measured GEP_{max} values are presented in Appendix B, Figure – B.2. It shows that most of

the data are within 50% quartile in 2009 (Appendix B: Figure –B.2a) and 2008 (Appendix B: Figure – B.2b) except for a few outliers.

4.1.2. Measured Respiration (R_{TOT})

The mean measured R_{TOT} , i.e. measured CO_2 at complete dark condition, at the cutover site was 12.718 ± 1.272 g CO_2 m^{-2} day^{-1} and 13.66 ± 1.95 g CO_2 m^{-2} day^{-1} in 2008 and 2009 respectively. At natural site it was 7.695 ± 0.55 in 2008 and 18.187 ± 2.44 g CO_2 m^{-2} day^{-1} in 2009. The measured mean R_{TOT} ranged from 7.14 ± 1.05 (Nhum) to 15.90 ± 2.14 (Car_aqu) g CO_2 m^{-2} day^{-1} in 2008 and 3.18 ± 0.29 (P) to 20.16 ± 4.92 (Cal_can) g CO_2 m^{-2} day^{-1} in 2009 (Appendix A: Table – A.1). One-way ANOVA test showed that there was significant difference of R_{TOT} between vegetation types in 2008 (ANOVA – $F = 21.77$, $P = 0.00$; Figure – 4.2) and 2009 (ANOVA – $F = 14.57$, $P = 0.00$; Figure – 4.2). It is important to note that in 2009 the natural vegetation (Nhum and Nhol) and Cal_can respired about two times more than 2008 while other communities showed similar respiration rate in both years of study (Appendix A: Table – A.1). Figure – B.3 (Appendix B) shows the distribution of all measured R_{TOT} in 2008 (Appendix B: Figure – B.3b) and 2009 (Appendix B: Figure – B.3a) where it shows that the data are fairly normally distributed and most of the data are within 50% quartile range with few outliers.

4.1.3. Maximum Net Ecosystem Exchange (NEE_{max})

The average maximum NEE (NEE at $PAR > 1000$ $\mu m^{-2} s^{-1}$) in 2008 was -13.66 ± 2.61 g CO_2 m^{-2} day^{-1} , -1.24 ± 1.19 g CO_2 m^{-2} day^{-1} and in 2009 -17.60 ± 3.84 g CO_2 m^{-2} day^{-1} , -12.38 ± 2.54 g CO_2 m^{-2} day^{-1} at recolonized areas of the cutover site and natural site, respectively. Negative values indicate uptake of CO_2 by the vegetation. There was a significant variation between vegetation types in average measured NEE_{max} , which is the sum

of GEP_{max} and R_{TOT} , in both 2008 (ANOVA – $F = 40.29$, $P = 0.00$; Figure – 4.3) and 2009 (ANOVA – $F = 21.34$, $P = 0.000$; Figure – 4.3). The lowest and highest mean NEE_{max} was observed at Nhum 0.003 ± 0.99 ($g\ CO_2\ m^{-2}\ day^{-1}$) and Sci_atr -25.58 ± 5.71 ($g\ CO_2\ m^{-2}\ day^{-1}$) in 2008 respectively (Appendix A: Table – A.1). The average NEE varied from 3.18 ± 0.29 $g\ CO_2\ m^{-2}\ day^{-1}$ at P to -22.78 ± 5.02 $g\ CO_2\ m^{-2}\ day^{-1}$ at Sci_atr in 2009 (Appendix A: Table – A.1). All vegetation had similar mean NEE during both years of study except Nhum which was several orders of magnitude higher, and Cal_can which was two times higher in 2009 than the previous year (Appendix A: Table – A.1). The Figure – B.4 (Appendix B) shows all measured NEE in 2008 (Appendix B: Figure – B.4b) and 2009 (Appendix B: Figure – B.4a).

4.2. Factors Controlling Measured CO_2 Exchange

4.2.1. Measured GEP_{max} Controlling Factors

Water table was significantly ($P < 0.05$) correlated with GEP_{max} for only Nhum and Nhol in 2008 but it was not a significant controlling factor for any vegetation type in 2009 (Table – 4.1). However, soil and air temperature had significant effect on GEP_{max} especially in 2009. The study found statistically significant correlation between temperature at 5cm depth and GEP_{max} for all vegetation communities in 2009 and for Equ_arv, Sci_atr, Eri_vag, and Nhol in 2008. In 2009 the mean soil temperature (average of 2, 5, 10, 15 and 20 cm depth) was also related to GEP_{max} for all vegetation types but only for Sci_atr, and Eri_vag in previous year. Similarly, average air temperature (temperature recorded from inside the chamber during CO_2 flux measurement) was also strongly correlated with GEP_{max} at all vegetation types in 2009 but only with Nhol in 2008. Like CH_4 flux, vegetation volume was also linked to CO_2 flux. The study found vegetation volume had significant effect on GEP_{max} for almost all vegetation except Eri_vag (Table – 4.1) in 2009.

4.2.2. Measured R_{TOT} Controlling Factors

Like GEP_{max} temperature and vegetation volume were the main controlling factors for respiration in 2009 and for few vegetation types in 2008. In general, water table was not an important control over R_{TOT} during the study period although a statistically significant correlation was found for a few vegetation types (Table – 4.2). Temperature at 5 cm depth was significantly correlated with R_{TOT} for all vegetation types except Cal_can in 2009, but it had control on R_{TOT} only at Equ_arv, Eri_vag and Nhum in 2008. In 2009, a similar result was observed for mean soil temperature. However, it was only significantly related with R_{TOT} at Car_aqu and Nhum. The mean air temperature was strongly correlated with R_{TOT} over the season at all vegetation types in 2009 but only at Car_aqu and Nhol in preceding year. Similar to GEP_{max} , vegetation volume had significant control over R_{TOT} for all vegetation types except Eri_vag and Nhum (Table – 4.2).

Although the study found similar environmental controls over measured GEP_{max} and R_{TOT} , it only found significant relation ($R^2 = 0.404$, $P < 0.001$) between GEP_{max} and R_{TOT} in 2008 for all vegetation types. In contrast no statistically relation was found in 2009 neither for all vegetation types together nor cutover and natural vegetation communities separately.

4.3. Modelled CO_2 Exchange

The study considered JD 131 – JD 243 (May 10 – Aug 31) of 2009 to calculate the modelled seasonal CO_2 fluxes. Although the study performed detailed process-based (e.g. CH_4 flux and its controlling factors, measured CO_2 flux and its controlling factor) research for two years, the modelling was done for only 2009 because of the availability of required data (e.g. vegetation volume and PAR, C fluxes measured in the early season).

4.3.1. Modelled GEP

GEP was modelled according to Equation-2.2. The values of model parameters i.e., GP_{max} , and *slopes* are presented in Table – 4.3 for all vegetation communities. The value of R^2 ranged from 0.552 to 0.892 (Table – 4.3). The comparison between measured and estimated modelled GEP to see the model fits are shown in Figure – B.5 (Appendix B). The modelled GEP varied significantly between the vegetation species at both mean daily and seasonal level (ANOVA – $F = 3.01$, $P = 0.025$; Mood's Median – Chi-square = 16.32, $P = 0.038$). The highest daily mean GEP was observed at Sci_atr followed by Eri_vag, Car_aqu, Cal_can, Typ_lat and Equ_arv at the cutover site. At the natural site it was Nhol>Nshrub>Nhum (Appendix A: Table –A.2). The seasonal total GEP ranged from $-1967.95 \pm 734.17 \text{ g CO}_2 \text{ m}^{-2}$ (Sci_atr) to $-865.69 \pm 202.27 \text{ g CO}_2 \text{ m}^{-2}$ (Nhum). The total seasonal value also showed the same pattern as daily mean GEP at both natural and cutover site (Table – 4.5).

The seasonal trend of modelled GEP over the growing season (Figure – 4.4) showed that for all vegetation types the GEP was very low at the beginning of the season and as the season progressed the GEP also increased until the third quarter of the season (JD 140 – JD 210) and after that it started to decrease again except Nhum (Figure – 4.4g) which showed almost level GEP throughout the growing season.

4.3.2. Modelled R_{TOT}

R_{TOT} was modelled using the equation 2.3. The model parameters are presented in Table – 4.4 for all vegetation types with respective R^2 value. The comparison between measured and estimated modelled R_{TOT} to see the model fits are shown in Figure – B.6 (Appendix B). The daily mean modelled R_{TOT} varied from $18.37 \pm 0.35 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Cal_can) to $1.65 \pm 0.18 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (P) at cutover site and $13.54 \pm 3.32 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$

(Nhum) to $10.65 \pm 0.46 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Nshrub) at natural site (Appendix A: Table –A.2). The highest seasonal R_{TOT} was observed at Cal_can and Nhum, and the lowest at P and Nshrub at cutover and natural sites, respectively (Table – 4.5). The study found statistically significant variation (ANOVA – $F = 14.10$, $P = 0.00$; Mood's Median – Chi-square = 24.33, $P = 0.002$) in daily mean and seasonal R_{TOT} between different vegetation types.

Unlike modelled GEP, modelled R_{TOT} did not show any regular pattern over the season. However, for most the vegetation types at the beginning of the season R_{TOT} was high and then there was a sharp fall around JD 140 except Cal_can and Typ_lat which followed the same pattern as modelled GEP. R_{TOT} then started to increase from about JD160 and reached a peak within 10 days and decreased over the next 10 days and levelled off again. Within couple of days it started to increase again and reached at maximum level point and maintains this level until the end of the season (Figure – 4.4). This up and down features in the seasonal R_{TOT} was mainly caused by the environmental parameters which controls CO_2 production i.e. water level and temperature.

4.3.3. Modelled NEE

The modelled NEE, which is the sum of modelled GEP and R_{TOT} , also varied significantly (ANOVA – $F = 14.10$, $P = 0.00$; Mood's Median – Chi-square = 16.32, $P = 0.038$) between different vegetation communities at natural and cutover site at both daily and seasonal level. At cutover site Car_aqu, Typ_lat, Sci_atr and Eri_vag, and at natural site Nhol and Nshrub were net sinks of C both daily and seasonally. Values ranged from -4.20 ± 1.38 (Eri_vag) to $-2.99 \pm 4.00 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Sci_atr) for daily average and -474.47 ± 155.79 (Eri_vag) to $-338.24 \pm 452.45 \text{ g CO}_2 \text{ m}^{-2}$ (Sci_atr) for seasonal total at cutover site (Appendix A: Table – A.2; Table – 4.5). At natural site modelled NEE varied from -3.94 ± 1.64 (Nhol) to

$-1.67 \pm 3.02 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Nshrub) for daily mean and -444.69 ± 185.29 (Nhol) to $-189.37 \pm 340.80 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Nshrub) for seasonal total.

The seasonal modelled NEE trend followed the GEP pattern at productive sites (Car_aqu, Typ_lat, Sci_atr, Eri_vag and Nhol). However, at less productive site (Cal_can, Equ_arv, Nhum and Nshrub) NEE followed the R_{TOT} pattern. On days when R_{TOT} was high, NEE tended to be positive i.e. source of CO_2 . However in general, NEE was close to zero at the beginning of the season and reached its maximum uptake point at the middle of the season following which it started decreasing again (Figure – 4.4).

4.4. Carbon Balance

The contribution of seasonal CH_4 efflux to the total C balance for cutover and natural vegetation communities and bare peat was very small compared to seasonal CO_2 flux (Figure – 4.5). The total C emitted from the bare peat was $50.97 \pm 5.63 \text{ g C m}^{-2}$ of which the share of $\text{CH}_4\text{-C}$ was almost zero ($0.01 \pm 0.03 \text{ g C m}^{-2}$). The only source of C for bare peat was $\text{CO}_2\text{-C}$ as respiration. Among the recolonized vegetation communities Cal_can ($156.61 \pm 38.81 \text{ g C m}^{-2}$) and Equ_arv ($75.21 \pm 29.23 \text{ g C m}^{-2}$) and at natural site Nhum ($181.32 \pm 46.36 \text{ g C m}^{-2}$) were net sources of C to the atmosphere (Table – 4.6). All other recolonized and natural vegetation was a net sink of C during the study, where total GEP exceeded total R_{TOT} and CH_4 flux. At the cutover site Eri_vag ($-126.60 \pm 42.96 \text{ g C m}^{-2}$) was the largest C sink followed by Car_aqu ($-109.24 \pm 24.29 \text{ g C m}^{-2}$), Typ_lat ($-101.29 \pm 13.85 \text{ g C m}^{-2}$) and Sci_atr ($-90.85 \pm 123.53 \text{ g C m}^{-2}$). Nhol stored more than two times more C than Nshrub during 2009 (Table - 4.6). Similar to P, for all the vegetation types the share of $\text{CH}_4\text{-C}$ to the total C balance was very low ($< 1\%$) compared to $\text{CO}_2\text{-C}$.

However, from global warming potential (GWP) perspective, CH₄ has more effect on total greenhouse gas balance. With the incorporation of CO₂ equivalent value for CH₄ flux as GWP, the result for total greenhouse gas balance changed compared to the C balance. Typ_{lat} which was a net C sink from a C perspective turned into an important source for CO₂-e ($211.91 \pm 298.03 \text{ g CO}_2\text{-e m}^{-2}$). Cal_{can} and Equ_{arv} at recolonized sites and Nhum at natural sites were also sources of greenhouse gases. All remaining vegetation types were greenhouse gas sinks from GWP point of view (Figure – 4.6).

4.5. Discussion on CO₂ Flux and its Controls

4.5.1. Measured CO₂ Flux

The GEP_{max} (PAR $\geq 1000 \mu\text{mol m}^{-2} \text{ s}^{-1}$), R_{TOT} and NEE_{max} vary both spatially between sites (Bubier *et al.*, 1998; Waddington and Warner, 2001) and year to year (Bubier *et al.*, 1998; Saarnio *et al.*, 2007). However, the average GEP_{max} value reported in rich fen is $-46.01 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Bubier *et al.*, 1998) and in minerotrophic peatlands is $-19.012 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Frolking *et al.*, 1998) which is much lower than for upland ecosystems (forest, grass land and cropland) where the value ranges from -134.23 to $-315.22 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Bubier *et al.*, 1998). The growing season mean R_{TOT} from many studies shows that it varies from 3.66 to $25.725 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Martikainen *et al.*, 1995; Alm *et al.*, 1997; Frolking *et al.*, 1998; Ikkonen *et al.*, 2001; Moore, 2002; Saarnio *et al.*, 2003) and there is almost no variation between different kinds of peatlands (Updegraff *et al.*, 2001; Moore *et al.*, 2002; Chimner and Cooper, 2003). The mean measured R_{TOT} for bare peat in both natural and laboratory condition is $5.2 \pm 4.2 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ (Waddington *et al.*, 2010). Bubier *et al.* (1998) found NEE_{max} at a dry site in rich fen of $-26.24 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ and at wet sites it varied from 1.90 to $7.6 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$. However, studies shows that the NEE_{max} for cutover and restored moss

sites is $2.28 \pm 6.99 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ and $-5.053 \pm 5.743 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$; and at cutover and restored herbaceous sites is $-14.65 \pm 1.25 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ and $1.3 \pm 0.65 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ respectively (Waddington *et al.*, 2010).

At BSF recolonized cutover site for all vegetation types GEP_{max} was much higher than the literature average reported for minerotrophic peatlands but less than rich fen except Cal_can ($-15.05 \pm 1.91 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) and Equ_arv ($-14.44 \pm 1.61 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) in 2008, and R_{TOT} was within the literature range. However, the mean measured R_{TOT} for bare peat ($3.18 \pm 0.29 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$) was lower than previously reported value. NEE_{max} of Sci_atr was similar to dry rich fen in both years of study. Car_aqu and Equ_arv in both years of study, and Cal_can, Typ_lat in 2008 showed NEE_{max} more than reported cutover herbaceous site but less than dry rich fen. Equ_arv in both years and Cal_can in 2008 showed lower NEE_{max} than cited cutover herbaceous site.

At BSF natural site GEP_{max} was lower than average minerotrophic peatlands for Nhum in both years of study and Nhol in 2008. However Nhol and Nshrub were higher than average minerotrophic peatlands but less than rich fen in 2009. Like the cutover site, R_{TOT} of natural vegetation plots was also within the average peatland range during both years of study. The NEE_{max} of Nhum was similar to reported cutover moss during this study. However, NEE_{max} of Nhol was and Nshrub in 2009 was similar to reported rich fen and cutover herbaceous site respectively.

The differences between GEP_{max} observed in the present study with reported values were mainly due to vegetation differences as photosynthesis of different species under same PAR is different. In some cases other environmental factors are also important for explaining the observed patterns. Many studies have found significant effect of vegetation biomass

(Bubier *et al.*, 1998; Komulainen *et al.*, 1999; Riutta *et al.*, 2007; Heikkinen *et al.*, 2002; Kivimäki *et al.*, 2008), soil (Bubier *et al.*, 1998; Komulainen *et al.*, 1999; Riutta *et al.*, 2007) and air temperature (Bubier *et al.*, 1998; Riutta *et al.*, 2007; Kivimäki *et al.*, 2008) and water table (Bubier *et al.*, 1998; Riutta *et al.*, 2007) on GEP_{max} . In this study vegetation volume was also an important controlling factor for measured GEP_{max} for all vegetation types except Typ_lat. However, this study did not find any significant water table correlation with GEP_{max} except in 2008 at natural vegetation. This is probably due to vegetation characteristics and deep water table conditions at BSF. Most of the studied vegetation communities at BSF consist of sedges and shrubs, except Nhum which was mainly moss, and previous studies found that sedges and shrubs are insensitive to water table position (Riutta *et al.*, 2007). However, I found important relation of GEP_{max} with temperature at 5 cm, mean soil temperature and air temperature in 2009 but only few species in 2008. Petrone *et al.* (2003) suggested that higher PAR, air and soil temperatures favour conditions for GEP_{max} . The only explanation for lack of relation between air and soil temperature and GEP_{max} in 2008 is that 2008 was drier than 2009 and less number of measurements was recorded in 2008. The effects of dry and wet cycles for plants have been well documented (McNeil and Waddington, 2003 and references there in). According to McNeil and Waddington (2003) drying and subsequent wetting not only increases respiration but also stops photosynthesis and the time required to return earlier C exchange depends on plant's desiccation tolerance.

Like GEP_{max} and CH_4 flux the control of vegetation on R_{TOT} is well documented in numerous studies (Bubier *et al.*, 2003a; Riutta *et al.*, 2007; Kivimäki *et al.*, 2008). At the BSF peatland, this research also found vegetation control on R_{TOT} at all vegetation types except Typ_lat and Eri_vag. Typ_lat had a strong relation between water table and R_{TOT} in 2009. On

the other hand the vegetation volume of Eri_vag did not change much over the growing season while R_{TOT} varied, and thus no relation was observed.

There is considerable debate about the relation between water table and R_{TOT} among peatland scientists. Some have suggested that water table has strong influence on R_{TOT} (Silvola *et al.*, 1996b; Oechel *et al.*, 1998; Bubier *et al.*, 2003b) while others have not observed this correlation (Bubier *et al.*, 1998; Updegraff *et al.*, 2001; Lafleur *et al.*, 2005). In this study the relationship between water table and R_{TOT} was observed for the natural site in both years, and only at Cal_can and Typ_lat in 2009 for the cutover site. However, this relationship was not observed at the cutover site in 2008. This may be because at natural site there was always continuous supply of fresh substrate thus changes in water table changes R_{TOT} as fresh substrate is exposed to oxic conditions and decomposes faster than old substrate (Lafleur *et al.*, 2005). At recolonized cutover site, vegetation volume was very high and possibly it masks the water table effect by contributing more autotrophic respiration compared to heterotrophic respiration. Lafleur *et al.* (2005) also suggested that “ R_{TOT} is greatest and sensitive to moisture changes in the uppermost portions of the peat profile”. As the year 2009 was wetter than 2008 it is possible that Typ_lat and Cal_can had water table fluctuation in the surface depths where most of the R_{TOT} occurred.

The strong soil temperature dependence of R_{TOT} has already been mentioned in many studies (Silvola *et al.*, 1996b; Bubier *et al.*, 1998; Waddington and Warner, 2001; Bubier *et al.*, 2003a; Petrone *et al.*, 2003). Several studies suggested near-surface temperature can predict more accurate R_{TOT} than mean soil temperature (Bubier *et al.*, 1998; Lafleur *et al.*, 2005). At BSF, this research found strong dependence of R_{TOT} on mean soil temperature at all vegetation types in 2009 except Cal_can, and at Car_aqu and Nhum in 2008. I also found

significant relation between soil temperature at 5 cm depth and R_{TOT} at all vegetation plots in 2009 except Cal_can, and at Equ_arv, Eri_vag and Nhum in 2008. Thus, this research agrees with the finding that near surface soil temperature is more appropriate than mean soil temperature. Relation between air temperature and R_{TOT} has been found in very few studies (Bubier *et al.*, 2003a; Marinier *et al.*, 2004). Surprisingly, at BSF this research found R_{TOT} and air temperature relation at all vegetation types in 2009, and Car_aqu and Nhol in 2008 where Car_aqu was negatively correlated.

In summary, from above comparative discussion between previously studied peatlands value for measured GEP_{max} , R_{TOT} and NEE_{max} this research shows that overall GEP_{max} was higher than reported value but R_{TOT} was similar to reported ranges. However, NEE_{max} for most of the recolonized species was higher than previously reported herbaceous cutover site. At BSF peatland both GEP_{max} and R_{TOT} was controlled mostly by vegetation volume and temperature but not water table.

4.5.2. Modelled CO₂ Flux

Limpens *et al.* (2008) reviewed a range of literature and found that the NEE for bogs around the world ranges from a large uptake of about $-330.075 \text{ g CO}_2 \text{ m}^{-2}$ to a large source of more than $366.7 \text{ g CO}_2 \text{ m}^{-2}$. In Canada, Lafleur *et al.* (2001) calculated an annual CO₂ uptake for a boreal bog of $-248 \text{ g CO}_2 \text{ m}^{-2}$. Another recent estimate of seasonal NEE at a raised bog in Canada ranged from -7 to $-411 \text{ g CO}_2 \text{ m}^{-2}$ (Roulet *et al.*, 2007). The annual NEE range for minerotrophic peatlands (fens) does not vary as much compared to bogs (Limpens *et al.*, 2008). Saarnio *et al.* (2007) reviewed a range of minerotrophic peatlands and found the annual NEE at oligotrophic fen is $-359 \text{ g CO}_2 \text{ m}^{-2}$; poor intermediate fen hummocks and hollows are $44.01 \text{ g CO}_2 \text{ m}^{-2}$ and $40.34 \text{ g CO}_2 \text{ m}^{-2}$ respectively; and at poor fen hummocks and hollows

are $370.42 \text{ g CO}_2 \text{ m}^{-2}$ and $-7.34 \text{ g CO}_2 \text{ m}^{-2}$ respectively. However, Waddington *et al.* (2002) found that cutover peatland is a large source of CO_2 to the atmosphere. In his study the NEE for young and old cutover sites in dry year was 363 and $399 \text{ g CO}_2 \text{ m}^{-2}$, and in wet year it was 88 and $112 \text{ g CO}_2 \text{ m}^{-2}$ respectively.

The seasonal total GEP and R_{TOT} vary significantly from site to site (e.g. Waddington and Warner, 2001) and year to year (e.g. Roulet *et al.*, 2007). Gorham (1995) calculated average annual GEP for northern peatlands of $-1085.58 \text{ g CO}_2 \text{ m}^{-2}$ and R_{TOT} of $913.20 \text{ g CO}_2 \text{ m}^{-2}$. Waddington and Roulet (2000) studied a boreal patterned peatland in Sweden and their estimate for seasonal GEP varied from $-600.37 \text{ g CO}_2 \text{ m}^{-2}$ to $-731.66 \text{ g CO}_2 \text{ m}^{-2}$ and for R_{TOT} ranged from $529.59 \text{ g CO}_2 \text{ m}^{-2}$ to $720.66 \text{ g CO}_2 \text{ m}^{-2}$.

At BSF peatland, the growing season modelled NEE varied from a net uptake of $-474.47 \pm 155.79 \text{ g CO}_2 \text{ m}^{-2}$ to a net source of $663.97 \pm 169.5 \text{ g CO}_2 \text{ m}^{-2}$ for individual vegetation types. Thus, the vegetation types Car_aqu, Typ_lat, Sci_atr, Eri_vag, Nhol and Nshrub are better from C sink perspective than cited intermediate and poor fens where it was a net source or little sink. Eri_vag showed the highest NEE during the study of $-474.47 \pm 155.79 \text{ g CO}_2 \text{ m}^{-2}$. This result falls within literature values where it ranged from $-370.42 \text{ g CO}_2 \text{ m}^{-2}$ to $-663.82 \text{ g CO}_2 \text{ m}^{-2}$ at *Eriophroum vaginatum* plots (Kivimäki *et al.*, 2008). The seasonal NEE for pure *Carex* plots varied from $-165.04 \text{ g CO}_2 \text{ m}^{-2}$ to $-359.415 \text{ g CO}_2 \text{ m}^{-2}$ (Kivimäki *et al.*, 2008). At BSF peatland, the seasonal NEE for Car_aqu was $-418.03 \pm 84.75 \text{ g CO}_2 \text{ m}^{-2}$ which is more than the literature cited value. This is likely because the water table did not change a lot during the study period at Car_aqu plots whereas it varied about 20 cm in the Kivimäki *et al.* (2008) study and vegetation cover decreases for *Carex* spp. under low water table condition. The value for hummock (Nhum) at BSF was twice the source of CO_2 than poor fen

hummocks in literature but ~15 times higher than intermediate fen hummocks. However, the hollows were several orders of magnitude greater sink of CO₂ than reported poor fen hollows. This is probably due to vegetation differences, longer growing season as well as possibly other environmental controls such as different water table positions and chemistry.

This current study found higher seasonal GEP at all vegetation plots than Swedish peatlands. The finding was also higher GEP at all vegetation plots than average northern peatlands except at Nhum and Equ_arv. This is because the studied Swedish peatland was mainly *Sphagnum* dominated (Waddington and Roulet, 2000) whereas BSF is mostly dominated by vascular vegetation which has higher productivity than *Sphagnum*. The modelled R_{TOT} was much higher than both Swedish and average northern peatlands at BSF. However, bare peat showed almost eight times less than previously studied seasonal emission in dry season and two times less in wet season at cutover site. In Canadian peatlands the greatest value of R_{TOT} observed at cutover site is 1459.66 g CO₂ m⁻² (Waddington and Warner, 2001). At BSF cutover recolonized peatlands the R_{TOT} value ranged from 939.88 ± 17.26 g CO₂ m⁻² to 2076.05 ± 379.15 g CO₂ m⁻². Among the cutover vegetation types Cal_can and Sci_atr showed higher R_{TOT} and all other vegetation showed lower value than the reported one. The seasonal R_{TOT} value for bare peat (186.91 ± 20.53 g CO₂ m⁻²) was almost seven times less than the above Canadian value.

Bubier *et al.* (1998) found similar pattern of GEP and R_{TOT} throughout the season where R_{TOT} increased as GEP increased and R_{TOT} decreased when GEP decreased. In this study the seasonal pattern of GEP and R_{TOT} was also consistent (Figure – 4.4) except during low water table conditions when R_{TOT} decreased (JD 170 - 180). At BSF, GEP and R_{TOT} were close to zero at the beginning of the season when the water in the soil was still frozen and

R_{TOT} quickly started to increase when the soil thawed. Thus this result is consistent with other studies (Bubier *et al.*, 1998; Waddington and Warner, 2001). The highest GEP and R_{TOT} were observed in the middle of the season as observed elsewhere (for example, Bubier *et al.*, 1998; Waddington and Warner, 2001). Although GEP decreased at the end of the season R_{TOT} was still high at the end of the season, which contradicts with literature. The hypothesis for this is that at the end of the season most of the vegetation died and thus GEP decreased however there was enough litter left, as well as soil temperature high enough to support the continuous R_{TOT} . This difference is likely because this study calculated the growing season only until August which was earlier than many reported C balances.

It was hypothesized that natural peatland will act as a CO_2 sink and bare peat will be a large source. This research found that natural hollow and shrub dominated sites were a net sink but hummock dominated sites were a source of CO_2 . However bare peat was a source but not as larger source of a source as reported in previous literature (e.g. Waddington *et al.*, 2002). It was also hypothesized that the spontaneously recolonized site may act as a CO_2 sink during the growing season if biomass is high, and that water table, soil and air temperature will control CO_2 flux. The results observed in this study were consistent with these hypotheses.

4.6. Discussion on Carbon Balance

Carbon balance is the total C budget of CO_2 -C, CH_4 -C, DOC and DIC for the whole year (Gorham, 1995; Waddington and Roulet, 2000). This study only considered CO_2 -C and CH_4 -C to calculate the C balance for the growing season. Roulet *et al.* (2007) suggested that overlooking CH_4 -C and DOC in the C balance results in an overestimation of C sink by -16 to -23 g C m⁻². However, Alm *et al.* (1999b) calculated that a growing season NEE of -64 to -76

g CO₂-C m⁻² is needed to support the annual C sink function of peatlands considering summer and winter R_{TOT} and CH₄ fluxes as well as annual leaching. One study found the contribution of CH₄-C and DOC/DIC were almost in equal amount (Waddington and Roulet, 2000) to the net ecosystem C balances (NECB) after NEE which contributes the major share in NECB (Limpens *et al.*, 2008). Alm *et al.* (1999a) found winter time R_{TOT}, CH₄ fluxes are 30 g CO₂-C m⁻² and 1 g CH₄-C m⁻² respectively in a European peatland and he concluded that this was similar to Canadian findings of Moore and Knowles (1990). The value of annual leaching C does not vary so much between Europe and Canada and ranges from 3.9 -7 g C m⁻² y⁻¹ (Alm *et al.*, 1999b and reference there in; Roulet *et al.*, 2007). The NECB for northern peatlands is quite conservative ranging from -10 to -30 g C m⁻² (Limpens *et al.*, 2008). The most cited NECB for northern peatlands is -23 g C m⁻² (Gorham, 1995). In this study Car_aqu, Typ_lat, Sci_atr, Eri_vag at recolonized site, and Nhol and Nshrub at natural site were net sinks of C during the growing season. The remaining species were net sources of C to the atmosphere. Among the C sink species Car_aqu, Typ_lat, Sci_atr were ~4 times, Eri_vag and Nhol, ~ 5 times, and Nshrub was twice as efficient C sink as the most cited average northern peatland value.

Inclusion of winter R_{TOT} and CH₄-C, and annual leaching as DOC/DIC must reduce the C sink function of the above C sink species. Taking into account all of the above literature cited values for winter R_{TOT} (30 g CO₂-C m⁻²) and CH₄-C (1 CH₄-C m⁻²), and annual maximum leaching flux (7 g C m⁻²) this research found that the C sink function will decrease 36%, 38%, 42%, 30%, 32% and 75% for Car_aqu, Typ_lat, Sci_atr, Eri_vag, Nhol and Nshrub respectively.

This study further investigated the greenhouse gas balance of the various vegetation communities by incorporating global warming potential (GWP) for CH₄ and converting fluxes into CO₂-e over the 100 years time horizon according to IPCC (2007). The main goal of this investigation was to see whether the studied vegetation types were greenhouse gas sources or sinks using standard IPCC accounting protocols. After incorporation of GWP, Typ_lat (211.91 ± 298.03 g CO₂-e m⁻²) became a net greenhouse gas source, even though it was a net sink of C (-101.285 ± 13.85 g C m⁻²) even ~four times more than the average northern peatlands. This was caused by higher CH₄ emissions compared to other vegetation types during this study. For all other vegetation types those that were C sinks were also greenhouse gas sinks, while C sources remained greenhouse gas sources. Inclusion of literature cited winter fluxes and annual leaching also decreased the green house gas sink functions of Car_aqu, Sci_atr, Eri_vag, Nhol and Nshrub by 25%, 22%, 17%, 16% and 38% respectively.

Table – 4.1: Pearson Correlation between GEP_{max} and Environmental Variables in 2008 and 2009.

Plot	W T (cm)		T_5 (°C)		T_{Soil} (°C)		T_{air} (°C)		V
	2008	2009	2008	2009	2008	2009	2008	2009	2009
Car_aqu	-0.037	-0.279	0.208	-0.565 ^a	-0.444	-0.577 ^a	0.528	-0.317 ^a	-0.404 ^a
Cal_can	-0.391	0.133	-0.308	-0.441 ^a	-0.169	-0.432 ^a	0.046	-0.388 ^a	-0.811 ^a
Equ_arv	-0.021	-0.104	-0.417 ^a	-0.741 ^a	-0.379	-0.751 ^a	0.109	-0.557 ^a	-0.765 ^a
Typ_lat	n.m.	0.016	n.m.	-0.605 ^a	n.m.	-0.638 ^a	n.m.	-0.523 ^a	-0.589 ^a
Sci_atr	0.153	0.045	-0.718 ^a	-0.576 ^a	-0.604 ^a	-0.537 ^a	0.014	-0.408 ^a	-0.721 ^a
Eri_vag	-0.316	0.169	-0.625 ^a	-0.518 ^a	-0.481 ^a	-0.511 ^a	0.172	-0.434 ^a	-0.181
Nhum	0.640 ^a	0.199	-0.386	-0.523 ^a	-0.323	-0.516 ^a	-0.150	-0.564 ^a	-0.403 ^a
Nhol	0.509 ^a	0.028	0.486 ^a	-0.698 ^a	0.376	-0.684 ^a	-0.497 ^a	-0.650 ^a	-0.672 ^a
Nshrub	n.m.	0.047	n.m.	-0.616 ^a	n.m.	-0.606 ^a	n.m.	-0.507 ^a	-0.464 ^a

1. n.m. – not measured.

2. ^a correlation is significant at $P < 0.05$.

3. V – Vegetation volume.

Table – 4.2: Pearson Correlation between R_{TOT} and Environmental Variables in 2008 and 2009.

Plot	W T (cm)		T_5 (°C)		T_{Soil} (°C)		T_{air} (°C)		V
	2008	2009	2008	2009	2008	2009	2008	2009	2009
Car_aqu	-0.078	0.085	-0.406	0.758 ^a	0.807 ^a	0.769 ^a	-0.879 ^a	0.742 ^a	0.338 ^a
Cal_can	0.336	-0.495 ^a	0.148	0.286	-0.003	0.281	-0.071	0.637 ^a	0.757 ^a
Equ_arv	-0.001	-0.083	0.413 ^a	0.710 ^a	0.360	0.734 ^a	-0.210	0.790 ^a	0.605 ^a
Typ_lat	n.m.	-0.517 ^a	n.m.	0.425 ^a	n.m.	0.409 ^a	n.m.	0.785 ^a	0.496 ^a
Sci_atr	0.091	-0.169	0.358	0.621 ^a	0.282	0.610 ^a	0.061	0.864 ^a	0.552 ^a
Eri_vag	0.260	-0.150	0.559 ^a	0.781 ^a	0.413	0.790 ^a	-0.210	0.674 ^a	0.230
P	n.m.	-0.091	n.m.	0.426 ^a	n.m.	0.404 ^a	n.m.	0.554 ^a	
Nhum	-0.610 ^a	-0.381 ^a	0.598 ^a	0.598 ^a	0.556 ^a	0.556 ^a	-0.036	0.605 ^a	0.138
Nhol	-0.535 ^a	-0.534 ^a	-0.058	0.866 ^a	-0.039	0.883 ^a	0.476 ^a	0.785 ^a	0.524 ^a
Nshrub	n.m.	-0.495 ^a	n.m.	0.742 ^a	n.m.	0.764 ^a	n.m.	0.633 ^a	0.373 ^a

1. n.m. – not measured.

2. ^a correlation is significant at $P < 0.05$.

3. V – Vegetation volume.

Table – 4.3: The value of GP_{max} , R^2 and slopes from GEP model (Model -2.2).

<i>Vegetation</i>	GP_{max}	R^2	Q	a
Car_aqu	-160.796	0.7517	-0.2132	0.0094
Cal_can	-93.108	0.8439	-0.1228	0.0705
Equ_arv	-65.4995	0.8676	-0.0960	0.1224
Typ_lat	-200.888	0.5897	-0.3522	0.0092
Sci_atr	-3639.48	0.7263	-7.5078	0.0006
Eri_vag	-57.8403	0.7065	-0.6918	0.1223
Nhum	-24.8508	0.5520	-0.0474	0.6194
Nhol	-69.9729	0.8924	-0.1035	0.2851
Nshrub	-73.9568	0.6636	-0.0911	0.0862

Table – 4.4: The value of R^2 , slopes and constant from Respiration model (Model – 2.3).

<i>Vegetation</i>	R^2	a_1	b_1	c
Car_aqu	0.58	1.02	0.03	-5.01
Cal_can	0.58	0.12	0.62	9.74
Equ_arv	0.54	0.74	0.27	-0.01
Typ_lat	0.24	0.22	0.18	1.39
Sci_atr	0.51	1.13	0.27	-7.36
Eri_vag	0.63	1.06	-0.20	-1.42
Nhum	0.35	1.08	0.05	-1.02
Nhol	0.76	1.71	-0.35	-10.12
Nshrub	0.74	1.69	-0.16	-10.21
Bare peat (P)	0.14	0.03	---	-0.14

Table – 4.5: Seasonal modelled GEP, R_{TOT} , and NEE for bare peat, recolonized and natural vegetation. Standard deviations (\pm SD) are shown in brackets and measured from three (four for bare peat) replicate plots.

Plot	GEP (gCO₂ m⁻²) Mean (\pmSD)	R_{TOT} (gCO₂ m⁻²) Mean (\pmSD)	NEE (gCO₂ m⁻²) Mean (\pmSD)
Car_aqu	-1511.88 (344.65)	1093.84 (258.77)	-418.03 (84.75)
Cal_can	-1504.78 (511.43)	2076.05 (379.15)	571.27 (140.51)
Equ_arv	-1041.28 (166.39)	1312.68 (69.31)	271.40 (103.46)
Typ_lat	-1383.25 (35.77)	939.88 (17.26)	-443.37 (20.34)
Sci_atr	-1967.95 (734.17)	1629.71 (281.72)	-338.24 (452.45)
Eri_vag	-1549.12 (96.50)	1074.65 (60.72)	-474.47 (155.79)
Nhum	-865.69 (202.27)	1529.67 (375.16)	663.974 (169.5)
Nhol	-1652.29 (139.06)	1207.60 (46.97)	-444.69 (185.29)
Nshrub	-1393.08 (288.57)	1203.719 (52.43)	-189.37 (340.80)
P	n.d. ¹	186.91 (20.53)	186.91 (20.53)

1. n.d.– not determined.

Table – 4.6: Net seasonal Carbon (C) balance for cutover and natural vegetation, and bare peat. Standard deviations (\pm SD) are shown in brackets and measured from three (four for bare peat) replicate plots.

Plot	C from CH ₄ (g C m ⁻²)	C from CO ₂ (g C m ⁻²)	Net C (g C m ⁻²)
	Mean (\pm SD)	Mean (\pm SD)	Mean (\pm SD)
Car_aqu	4.74 (1.18)	-113.98 (23.11)	-109.24 (24.29)
Cal_can	0.84(0.50)	155.77(38.31)	156.61(38.81)
Equ_arv	1.21(1.02)	74.00(28.21)	75.21(29.23)
Typ_lat	19.61(8.31)	-120.89(5.55)	-101.29(13.85)
Sci_atr	1.37(0.16)	-92.23(123.37)	-90.85(123.53)
Eri_vag	2.78(0.48)	-129.37(42.48)	-126.60(42.96)
P	0.01(0.03)	50.96(5.60)	50.97(5.63)
Nhum	0.28(0.15)	181.04(46.22)	181.32(46.36)
Nhol	1.32(0.39)	-121.25(50.52)	-119.93(50.92)
Nshrub	0.58(0.43)	-51.63(92.92)	-51.06(93.36)

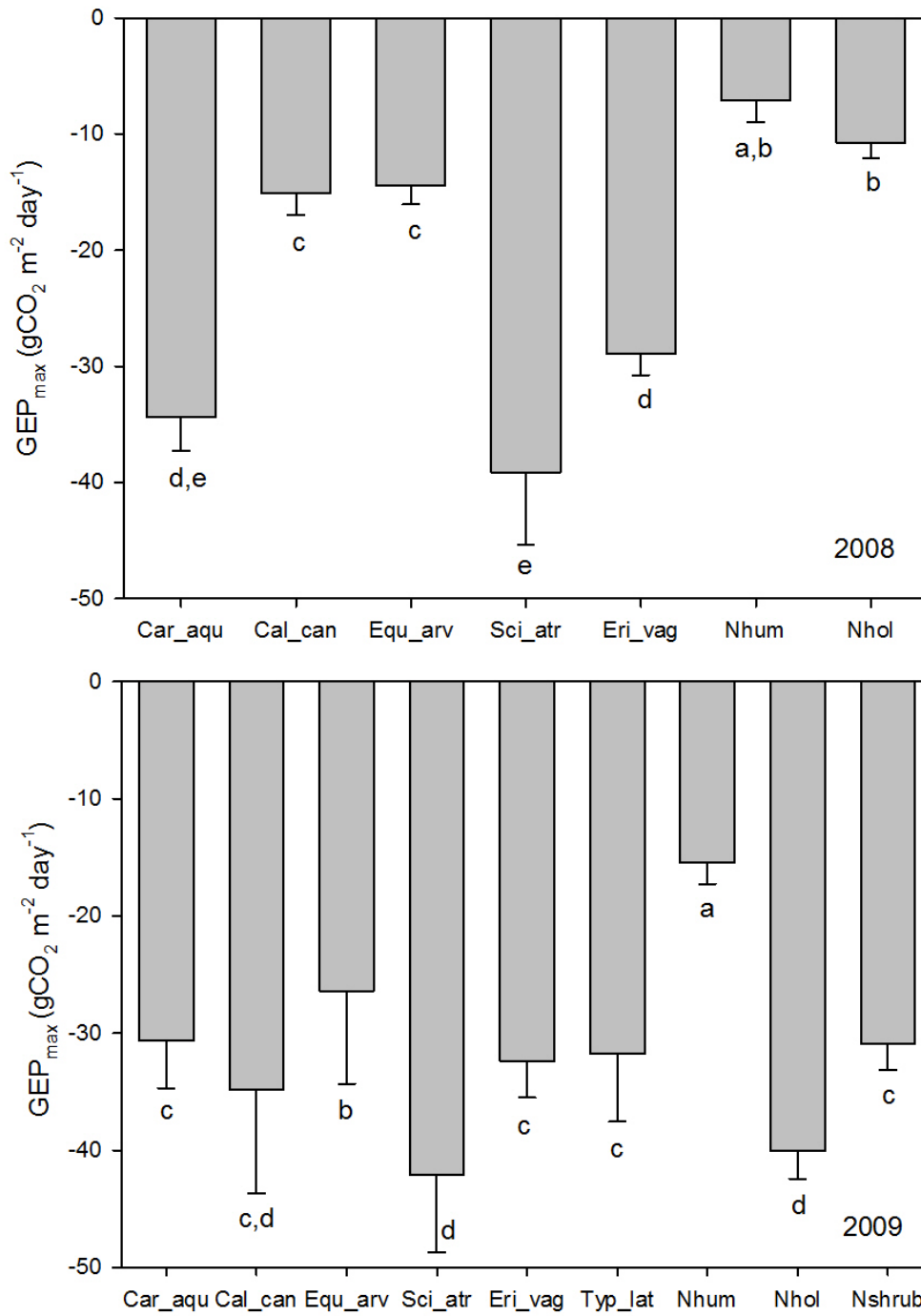


Figure – 4.1: Mean measured GEP_{max} in 2008 and 2009. Error bars are given standard deviation of mean value. Plot types are significantly different (P < 0.001) from each other if no letters are in common. Letters should be compared only within one plot.

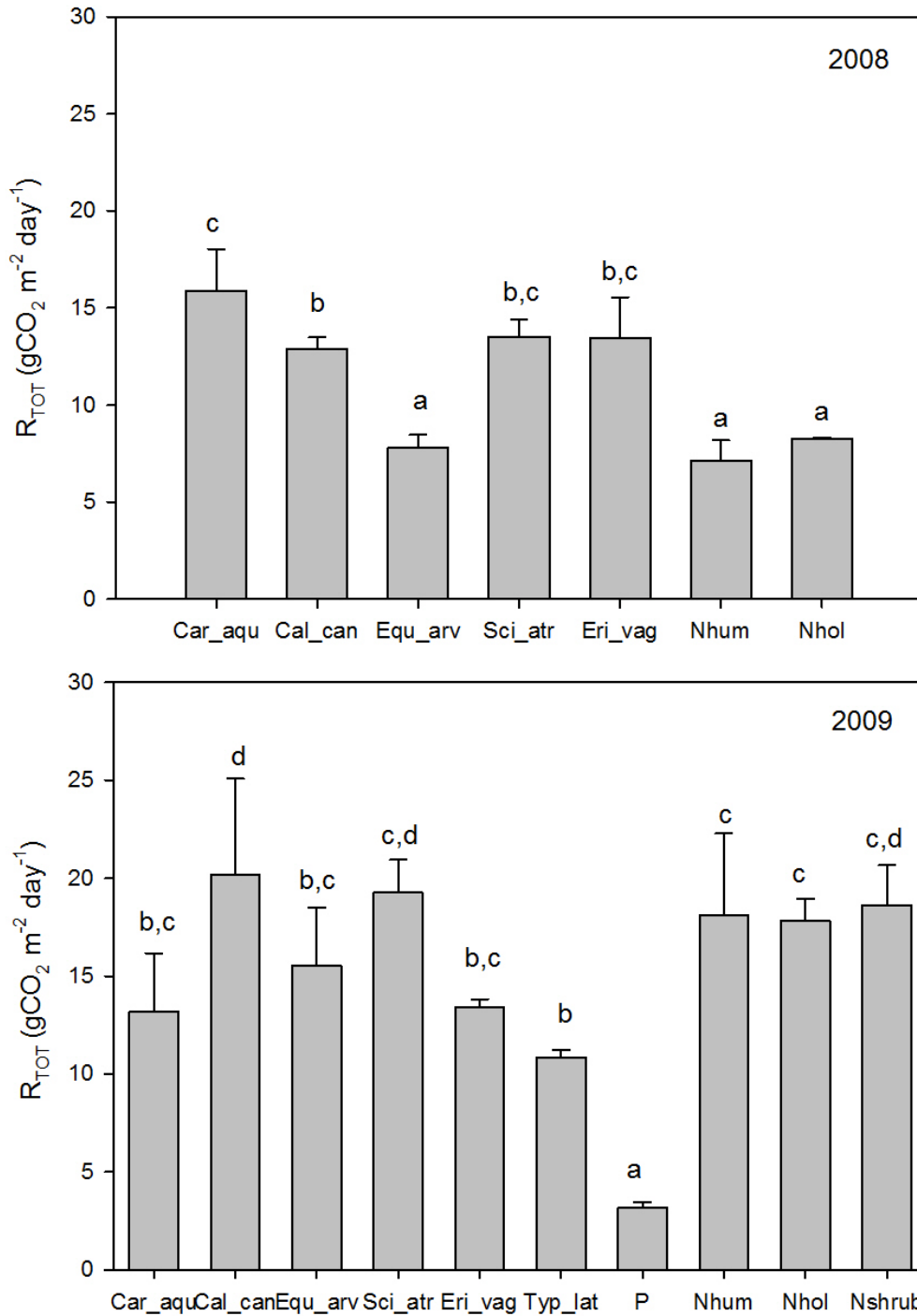


Figure – 4.2: Mean measured R_{TOT} in 2008 and 2009. Error bars are given standard deviation of mean value. Plot types are significantly different ($P < 0.001$) from each other if no letters are in common. Letters should only be compared within one plot.

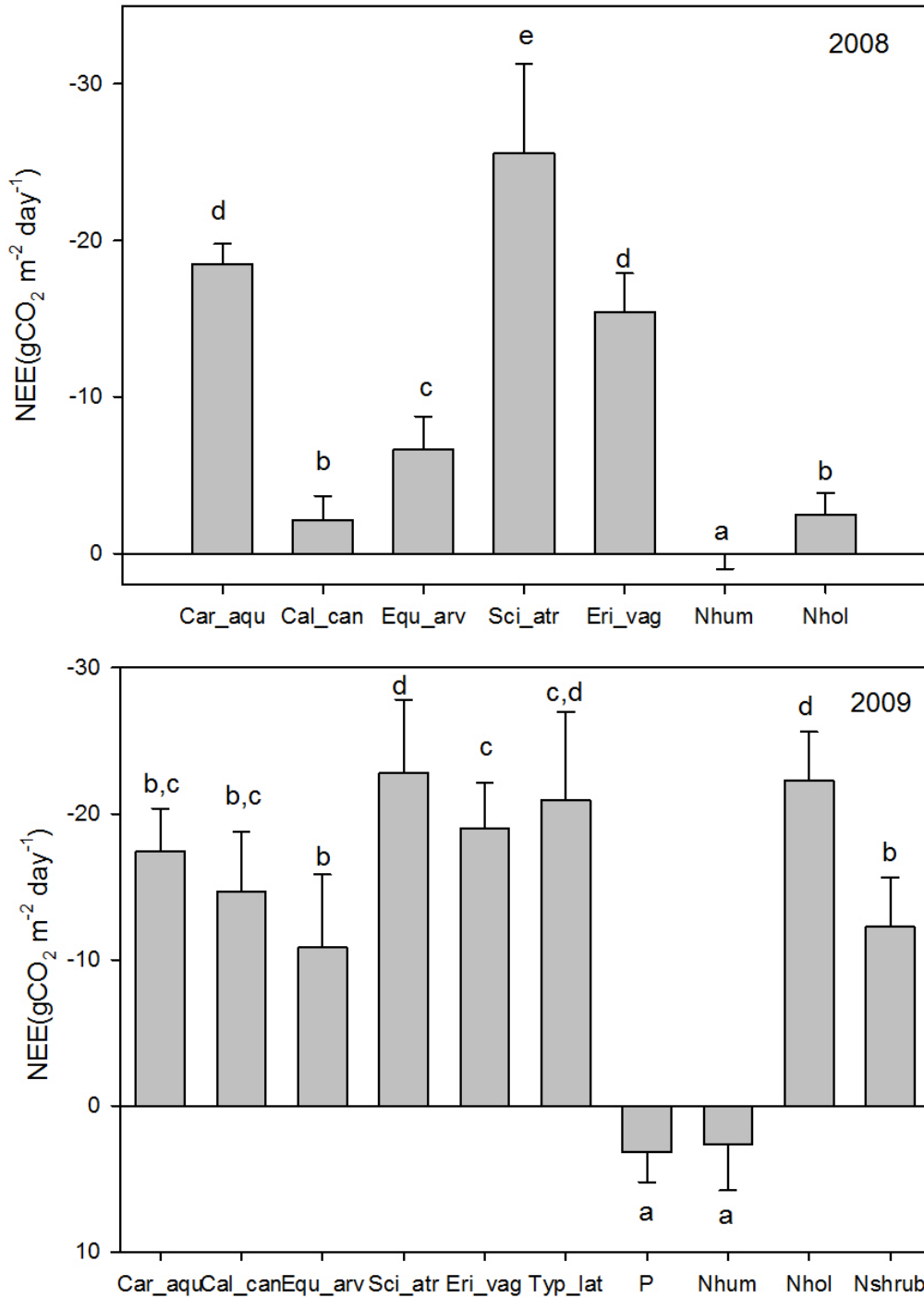
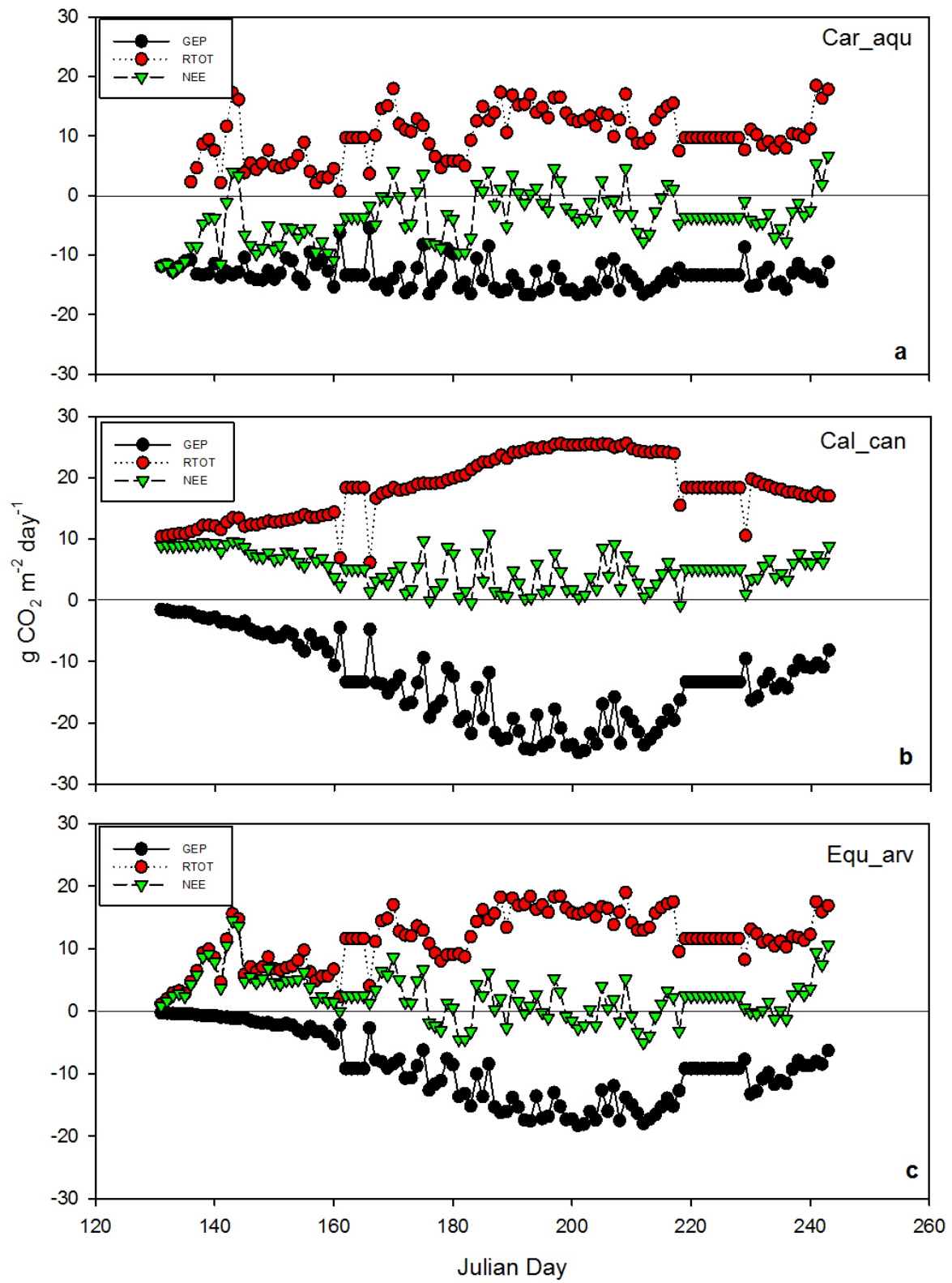
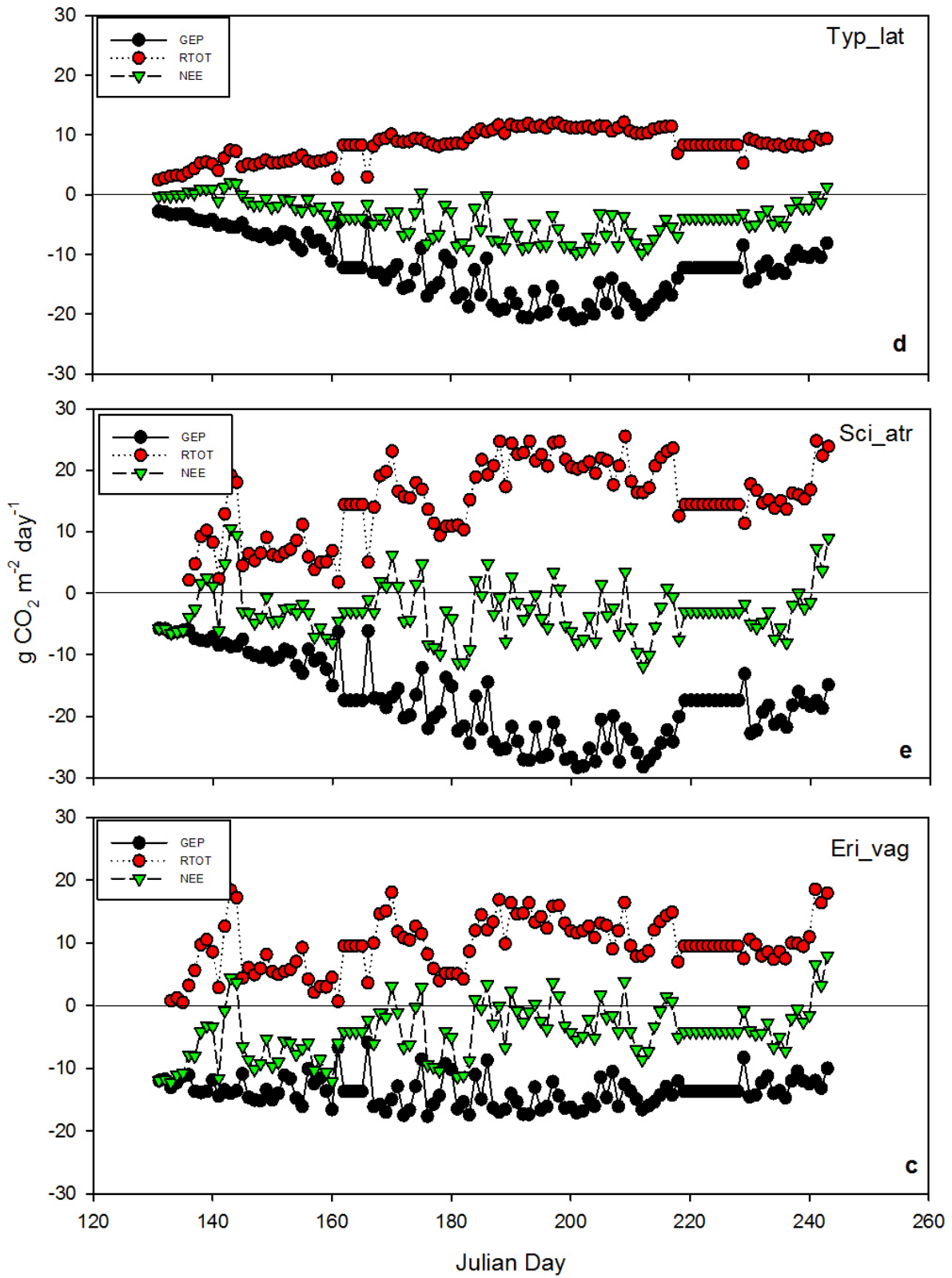


Figure – 4.3: Mean measured NEE in 2008 and 2009. Error bars are given standard deviation of mean value. Plot types are significantly different ($P < 0.001$) from each other if no letters are in common. Letters should only be compared within one plot.





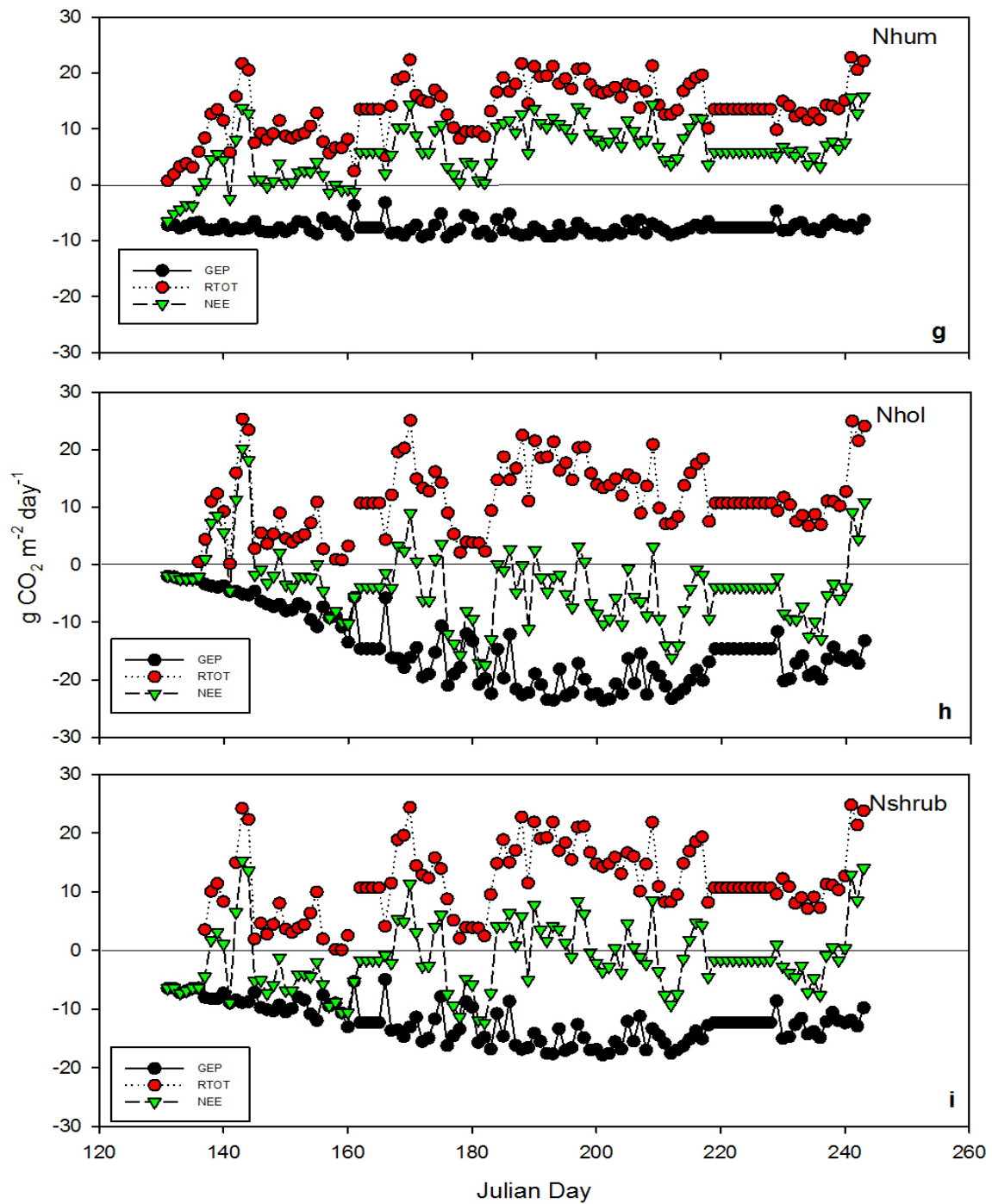


Figure – 4.4: Growing season modelled GEP, R_{TOT} and NEE from JD 131 – 243 all vegetation types (a-i). Any negative value indicating CO_2 sink and positive as source from the vegetation plots.

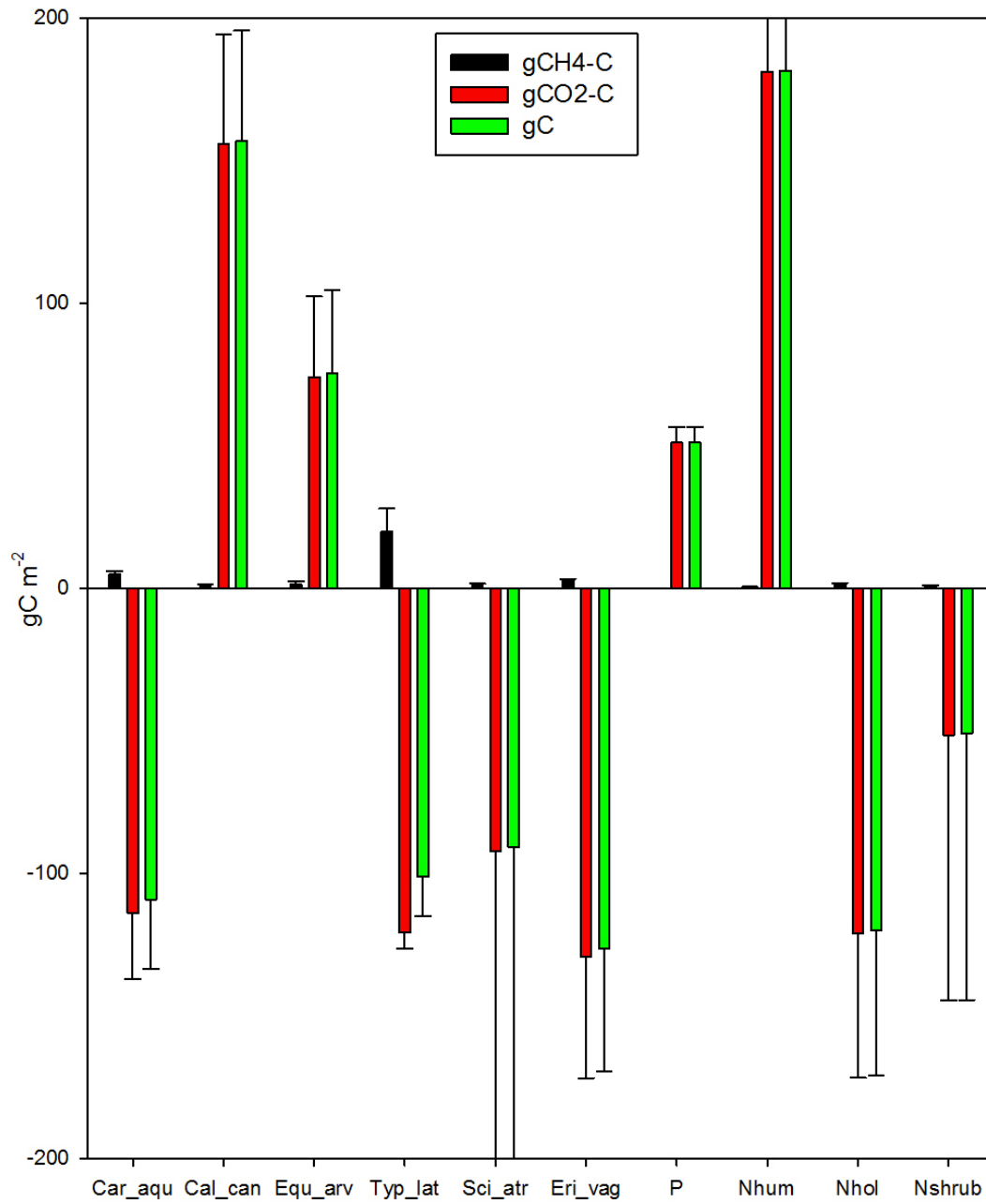


Figure – 4.5: Seasonal carbon balance for all vegetation types and bare peat with their $\text{CO}_2\text{-C}$ and $\text{CH}_4\text{-C}$ contribution. Error bars are showing standard deviation for replicate plots.

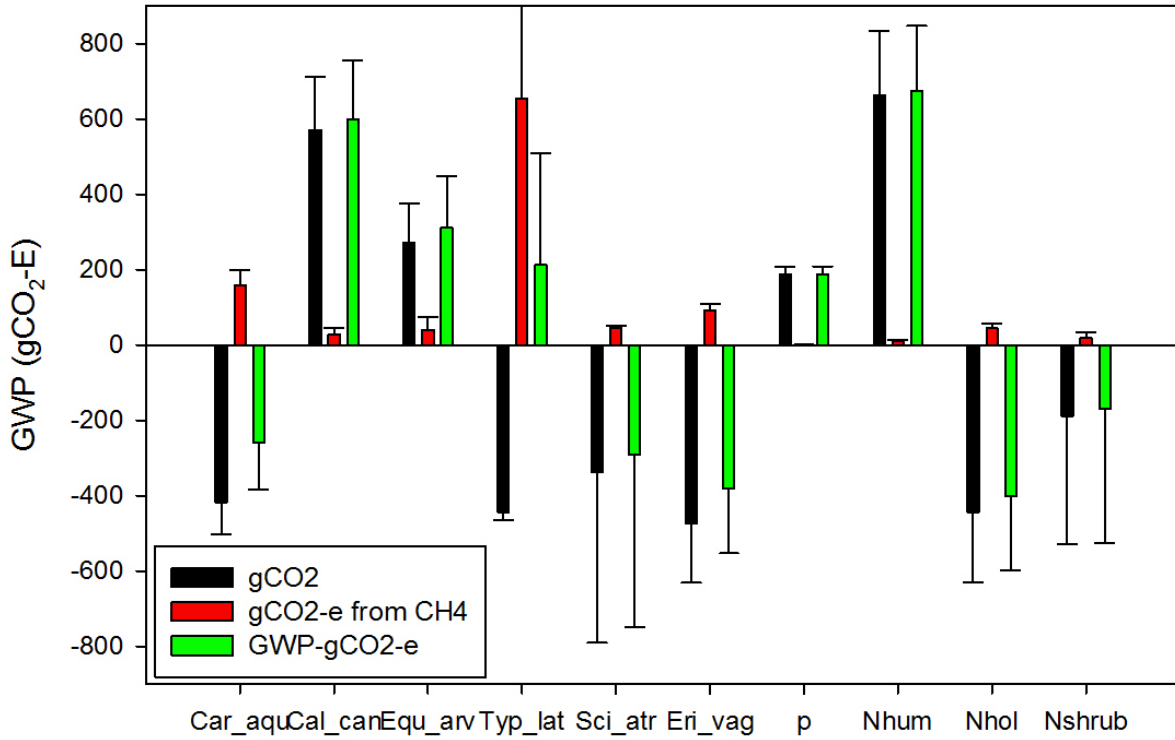


Figure – 4.6: Seasonal carbon balance with GWP in CO₂-e. Error bars are showing standard deviation of three replicate plots calculated for three replicate plots for all vegetation plots and four replicate plots for bare peat. The GWP for CH₄ was calculated multiplying the total seasonal CH₄ balance by 25 as CH₄ has GWP of 25 CO₂-e over 100 years time frame (IPCC, 2007).

CHAPTER FIVE: IMPLICATIONS FOR RESTORATION AND CONCLUSIONS

5.1. Summary

Mean \pm standard deviation 2009 growing season CH₄ emissions were 203.1 ± 109.5 to $978.9 \pm 291.5 \mu\text{mol m}^{-2} \text{ day}^{-1}$ (or 0.003 ± 0.002 to $0.016 \pm 0.005 \text{ g CH}_4 \text{ m}^{-2} \text{ day}^{-1}$) from vegetation communities of the undisturbed fen and $43.11 \pm 128.7 \mu\text{mol m}^{-2} \text{ day}^{-1}$ (or $0.0007 \pm 0.002 \text{ g CH}_4 \text{ m}^{-2} \text{ day}^{-1}$) from bare peat on the cutover site. Revegetation of the cutover site increased mean CH₄ flux to 621.8 ± 369.6 to $14497 \pm 6143 \mu\text{mol m}^{-2} \text{ day}^{-1}$ (or 0.010 ± 0.006 to $0.232 \pm 0.098 \text{ g CH}_4 \text{ m}^{-2} \text{ day}^{-1}$), although seasonal efflux varied significantly between vegetation types. The highest fluxes from recolonizing vegetation communities were observed for those dominated by *Typha latifolia*, *Carex aquatilis* and *Eriophorum vaginatum*. For the first two community types high rates of CH₄ efflux are likely linked to shallow water table position, however, for the latter, water table was on average greater than 20 cm below the surface and plant-mediated transport of CH₄ was clearly important for maintaining high efflux. In general, the combination of hydrological and ecological controls was important for controlling CH₄ flux across the site. The observation of CH₄ in pore water across all sampling locations suggests that CH₄ is being produced at the cutover peatland and that low measured fluxes likely result from substantial oxidation of CH₄ in the unsaturated zone.

Under full light conditions (PAR > 1000 $\mu\text{mol m}^{-2} \text{ s}^{-1}$) the measured maximum net ecosystem exchange (NEE_{max}) is the total of maximum ecosystem productivity (GEP_{max}) and total respiration (R_{TOT}). At cutover vegetation communities (NEE_{max}) measured mean \pm standard deviation was $-13.66 \pm 2.61 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ and $-17.60 \pm 3.84 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ in the 2008 and 2009 growing seasons respectively. In contrast, at natural site the NEE_{max} was $-1.24 \pm 1.19 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ and $-12.38 \pm 2.54 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$. Thus, revegetation increased

NEE_{max} at cutover site. The major controlling factors for GEP_{max} and R_{TOT} were vegetation volume and temperature in this study. The growing season modelled NEE varied from a net uptake of -474.47 ± 155.79 g CO₂ m⁻² to a net source of 663.97 ± 169.5 g CO₂ m⁻² for individual vegetation types. From the C sink point of view the species Car_aqu, Typ_lat, Sci_atr, Eri_vag, Nhol and Nshrub were higher than previously reported values for peatlands.

Total annual C balance for an ecosystem consists of annual CO₂ flux, CH₄ flux, and DOC/DIC leaching. The growing season total C balance at BSF peatlands (sum of CO₂ and CH₄ as gram carbon) found that Car_aqu, Typ_lat, Sci_atr, Eri_vag, Nhol and Nshrub were net sinks of C. This study also incorporated estimates of winter time fluxes and annual leaching from previous studies and found that the C sink function decreased by 30 to 75%. Further investigation including GWP for greenhouse gas balance showed that Typ_lat is a net source of greenhouse gas while the other C sink species were also greenhouse gas sinks. When this study added winter fluxes and annual leaching in to the GWP calculation, it shows that their greenhouse gas sink decreased from 16 to 38%.

Understanding community functions of spontaneously recolonizing species on cutover fens can help us to make decisions about the inclusion of these communities for future restoration measures. Although this study considered only growing season CO₂ and CH₄ fluxes, these results are valuable for restoration planning.

5.2. Implications for Restoration

It has been observed that vascular vegetation can assist in the establishment of bryophytes on cutover peat by improving microclimatic conditions (Graf and Rochefort, 2010) and thus maintaining these species that have spontaneously recolonized the site may be advantageous. Although this research did not measure directly wintertime CO₂ and CH₄ flux,

and the annual leaching of DOC/DIC, it considered the highest reported value to compare the difference in C balance. It further investigated the GWP for measured fluxes alone as well as incorporating above units. However, before making any decision on inclusion of species in the restored site species pool it is critical to determine other community functions such as invasive characteristics, eco-hydrological conditions of their successfully established communities and interaction with bryophyte species. Keeping this in mind, the suitability of the investigated species based on total C balance can be considered.

Among the recolonizing vegetation communities, Cal_can (*Calamagrostis canadensis*) and Equ_arv (*Equisetum arvense*) showed a net source of C to the atmosphere based on total C balance. When this study considered GWP for greenhouse gases these species become a greater source. Thus these species are undesirable in restoration from both a carbon accumulation and greenhouse gas exchange perspective.

Car_aqu (*Carex aquatilis*), Typ_lat (*Typha latifolia*), Sci_atr (*Scirpus atrocinctus*), and Eri_vag (*Eriophorum vaginatum*) were net sinks of C. However when this study incorporated GWP for greenhouse gases Typ_lat became net source, while the other species sink function decreased slightly. This is because Typ_lat released a large amount of CH₄. It grows only in ditches at BSF, a location known to have high CH₄ flux in cutover peatlands due to permanently inundated conditions (Sundh *et al.*, 2000; Waddington and Day, 2007). The presence of *Typha latifolia* likely increases CH₄ flux from the ditches beyond what would be observed from flooded locations alone due to its ability to provide labile substrate and to transport CH₄ from the soil to the atmosphere (Chanton *et al.*, 1992). Moreover, it is highly invasive in nature especially for wetlands (Shih and Finkelstein, 2008). Restoration of peatlands soon after abandonment may help to reduce the abundance of this species. The

vegetation type Eri_vag (*Eriophorum vaginatum*) has a deep root system, which vents methane from the anoxic peat layer allowing this vegetation type to emit significantly more CH₄ than other recolonizing species. However, it has a higher C accumulation function (as CO₂) especially during midsummer (Kivimäki *et al.*, 2008) and thus when total C balance was considered it was a net sink of C and even for greenhouse gases. Additionally this species is known for its prolific establishment in disturbed peatlands as the dominant vegetation (Malmer, 1986; Lavoie *et al.*, 2005; Tuittila *et al.*, 1999; Frenzel and Karofeld, 2000). When considering CH₄ emission from the fen this species is undesirable, however it has been shown to act as a nurse species for moss establishment (e.g. Tuittila *et al.*, 2000a) and the relative importance of these functions must be considered.

The vegetation type Car_aqu (*Carex aquatilis*) also releases significantly greater CH₄ than other species although this is clearly linked to its wet habitat preference. However this species was a net C sink from total C balance and greenhouse perspective because of high C accumulation capacities as CO₂. Moreover, this species has been used in European restoration because of its dispersal limited characteristics, and therefore limited invasive capabilities (Graf *et al.*, 2008). Despite high CH₄ emissions, in North America it may be considered for the restored species pool as additional services provided by this species are advantageous such as higher sequestration of CO₂ than average CH₄ emission. Thus it is likely acceptable for restoration purposes, especially as it is a dominant species in many natural fens in North America (Graf *et al.*, 2008).

Vegetation type Sci_atr (*Scripus atrocinctus*) was a net sink of C and greenhouse gases during this study and there are no reported disadvantages to ecological function for this species. Moreover Graf (2008) found *Scirpus cyperinus* had good organic matter accumulation

potential and that it could act as a nurse plant for mosses. Thus, we should considering including this species in restoration plans.

Generally during restoration the surface is scraped and then donor material is spread. It involves higher cost as well as disturbing other natural peatlands to collect donor materials. Moreover, Waddington *et al.* (2010) also found that following the restoration the C sink function does not improve a lot due to less vegetation cover. Thus if we can keep the existing C sink species and reintroduce additional plants then our goal of C sink may be achieved.

However, there are some questions unanswered in this research as it was conducted only at the plot scale within specific vegetation communities. Most of these unaddressed questions are related to larger ecosystem level and ecosystem complexities which are discussed below. The research needed to address these questions is discussed in the Future Work section (5.3).

Firstly, this research did not address what will be the C balance of specific communities (e.g. *Carex aquatilis*, *Scirpus atrocinctus*, *Eriophorum vaginatum*) for larger scale compared to this research plot scale. This research found that most of the studied vegetation grows as patches and assumed that the C balance will be the same in these patches as that measured in the study plots. However, there may be some changes in C balance if they will grow on a large scale as it will be affected by changed environmental conditions such as water table and soil characteristics.

Secondly, this research did not investigate the effect of different species associations on C balance. Previous studies have shown that C sinks function of sedges and *Sphagnum* for cutover peatlands increased when the diversity was high (Kivimäki *et al.*, 2008). There is also

unaddressed question of how the above suggested species will interact with each other and ultimately affect the C dynamics and peat accumulation characteristics.

Thirdly, restoration will require rewetting the site. This study did not answer the C dynamics under this changed condition for specific vegetation communities or for a vegetation complex of suggested species.

Thus, before taking any final decision to incorporate suggested species it is very crucial to address these above mentioned scenarios. However, it would be useful to start restoration work as soon as possible after harvesting of peatlands to avoid introduction of unwanted species into the site as well as severe damage to the system by compaction and erosion of peat.

5.3. Future Work

This study considered two growing seasons and measured CH₄ and CO₂ flux in a cutover peatlands. However, it performed CO₂ modelling for only 2009. It also calculated C balance for one growing season in 2009. It would be useful to study at least three years and calculate C balances for those years to see how C balance varies temporally as previous studies (e.g. Shannon and White, 1994) suggested that more than one year is required to get successful C flux of a peatland. Although it considered previously cited highest values for winter time fluxes and annual leaching, it would be useful to study these and incorporate them into the annual C balance estimation.

Restoration of peatlands requires rewetting of the site. Tuittila *et al.* (1999) found that rewetting for restoration lowered R_{TOT} and increased GEP and thus increased total C accumulation for *Eriophorum vaginatum*. Understanding how rewetting affects the C balance for all species alone and complex of the species considered in the present study would be

useful. Moreover, study of changes in C balance on a larger scale for individual vegetation and a combination of different species would be useful. It would also be useful to determine the survival of different species in a complex vegetation system and environmental conditions.

Further investigation incorporating these species into the current North American restoration method (rewetting and diaspore introduction) and seeing how this affects overall C accumulation and its trajectory following restoration will be useful for understanding whether spontaneously recolonizing species should be included in fen restoration procedures.

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APPENDICES

APPENDIX-A: ADDITIONAL TABLES

Table – A.1: Measured CO₂ flux (gCO₂ m⁻² day⁻¹) in 2008 and 2009. GEP was measured at ≥1000 PAR (μmol m⁻² s⁻¹) and R_{TOT} was measured at dark condition. Plus minus standard deviations are showing within the bracket.

Plot	2008			2009		
	<i>GEPmax</i> <i>Mean(±SD)</i>	<i>R_{TOT}</i> <i>Mean(±SD)</i>	<i>NEEmax</i> <i>Mean(±SD)</i>	<i>GEPmaxM</i> <i>ean(±SD)</i>	<i>R_{TOT}</i> <i>Mean(±SD)</i>	<i>NEEmax</i> <i>Mean(±SD)</i>
Car_aqu	-34.40(2.90)	15.90(2.14)	-18.49(1.28)	-30.65(4.05)	13.21(2.97)	-17.44(2.91)
Cal_can	-15.05(1.91)	12.90(0.58)	-2.15(1.52)	-34.85(8.81)	20.16(4.92)	-14.68(4.09)
Equ_arv	-14.44(1.61)	7.80(0.66)	-6.64(2.11)	-26.39(7.94)	15.54(2.97)	-10.85(4.97)
Typ_lat	n.m.	n.m.	n.m	-31.79(5.77)	10.86(0.37)	-20.93(6.04)
Sci_atr	-39.12(6.24)	13.53(0.88)	-25.58(5.71)	-42.04(6.64)	19.26(1.69)	-22.78(5.02)
Eri_vag	-28.90(1.86)	13.46(2.10)	-15.44(2.45)	-32.36(3.16)	13.39(0.44)	-18.97(3.18)
P	n.m.	n.m.	n.m	--	3.18(0.29)	3.18(0.29)
Nhum	-7.14(1.86)	7.14(1.05)	0.003(0.99)	-15.48(1.79)	18.09(4.20)	2.61(3.81)
Nhol	-10.73(1.35)	8.25(0.05)	-2.47(1.39)	-40.09(2.34)	17.83(1.10)	-22.26(3.35)
Nshrub	n.m.	n.m	n.m.	-30.91(2.26)	18.64(2.02)	-12.27(0.47)

Table – A.2: Daily mean modelled GEP, R_{TOT}, and NEE for bare peat, recolonized and natural vegetation. Plus minus standard deviations are showing within the bracket.

Plot	GEP (gCO ₂ m ⁻² day ⁻¹) Mean (±SD)	R _{TOT} (gCO ₂ m ⁻² day ⁻¹) Mean (±SD)	NEE (gCO ₂ m ⁻² day ⁻¹) Mean (±SD)
Car_aqu	-13.38 (3.05)	9.68 (2.29)	-3.70 (0.75)
Cal_can	-13.31 (4.53)	18.37 (3.35)	5.06 (1.24)
Equ_arv	-9.21 (1.47)	11.62 (0.61)	2.40 (0.92)
Typ_lat	-12.24 (0.32)	8.32 (0.15)	-3.92 (0.18)
Sci_atr	-17.42 (6.50)	14.42 (2.49)	-2.99 (4.00)
Eri_vag	-13.71 (0.85)	9.51 (0.54)	-4.20 (1.38)
Nhum	-7.66 (1.79)	13.54 (3.32)	5.88 (1.52)
Nhol	-14.62 (1.23)	10.69 (0.42)	-3.94 (1.64)
Nshrub	-12.33 (2.55)	10.65 (0.46)	-1.67 (3.02)
P	n.d.	1.65 (0.18)	1.65 (0.18)

n.d. – not determined

APPENDIX – B: ADDITIONAL FIGURES

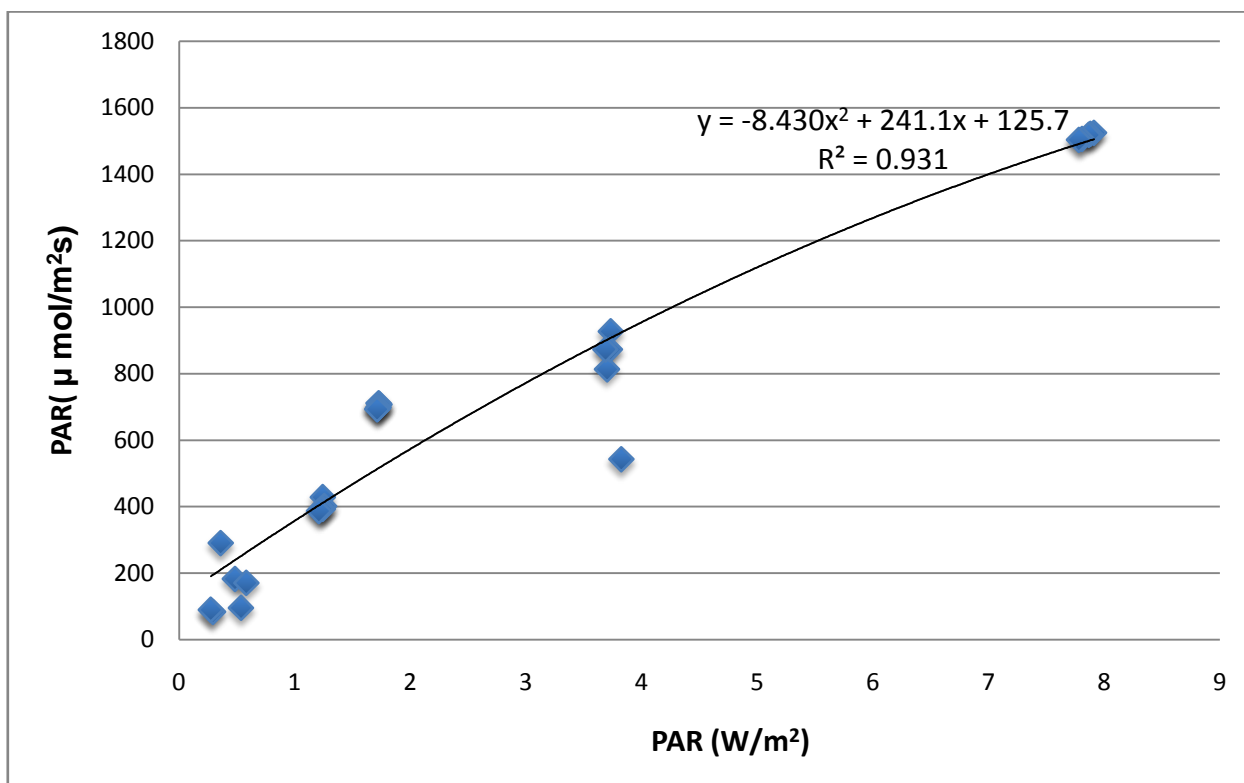


Figure – B.1: Relation between PAR measured at logger (W m^{-2}) at meteorological station and at PP system ($\mu \text{mol m}^{-2} \text{s}^{-1}$)

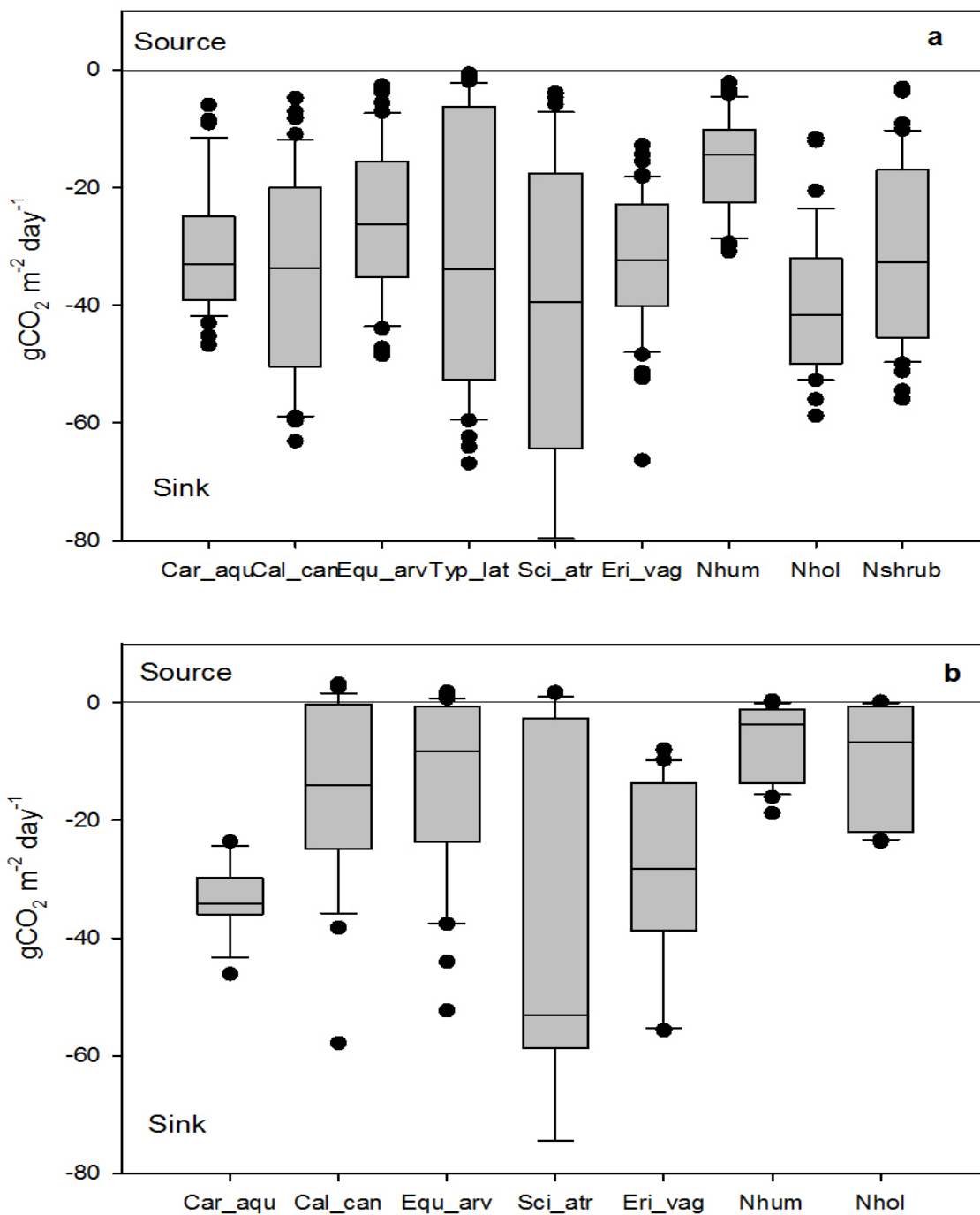


Figure – B.2: The distribution of all measured GEP_{max} in 2009 (a) and 2008 (b). Any positive value is showing source of CO_2 and negative showing sink of CO_2 .

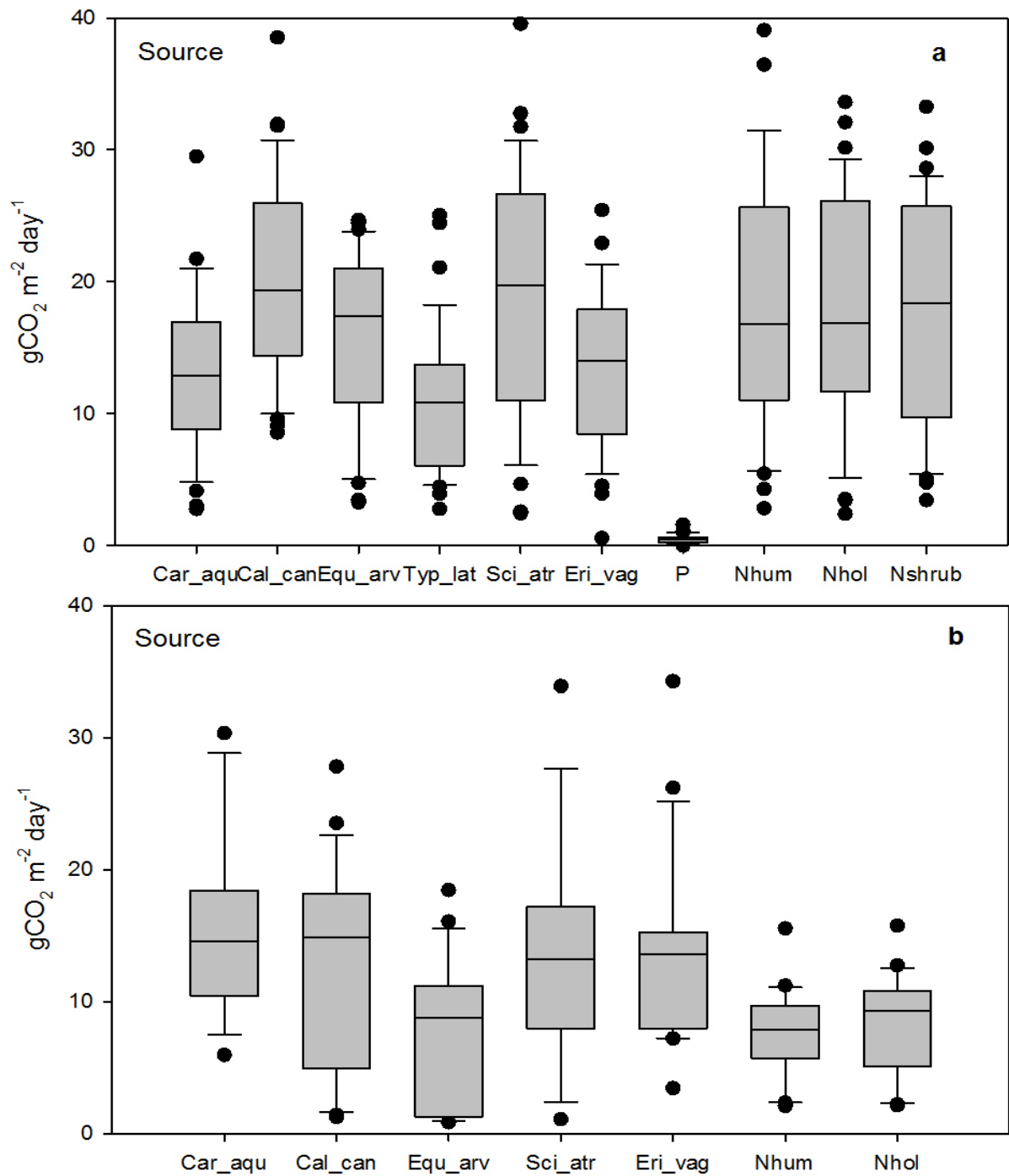


Figure – B.3: The distribution of all measured R_{TOT} in 2009 (a) and 2008 (b). Any positive value is showing source of CO_2 and negative showing sink of CO_2 .

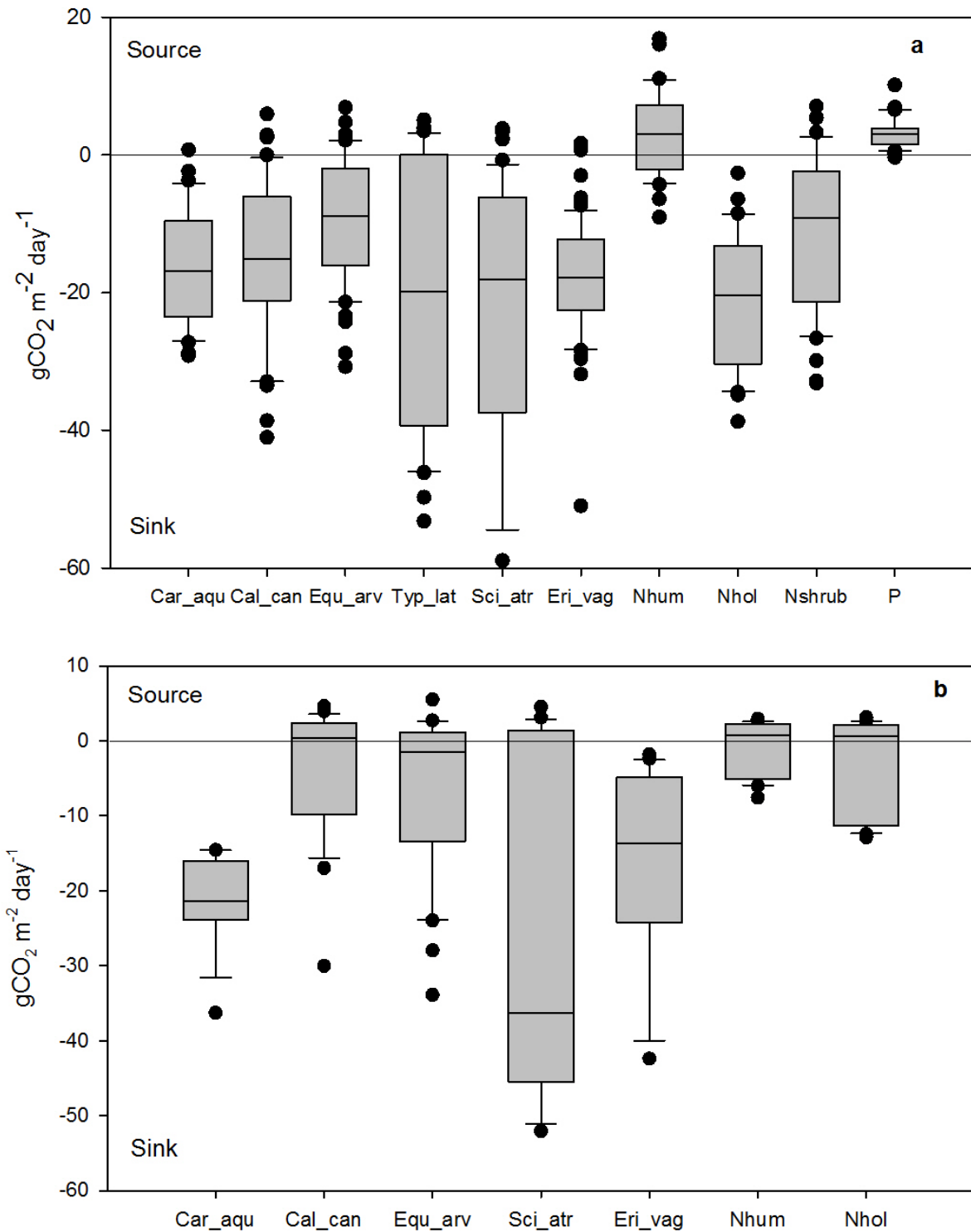
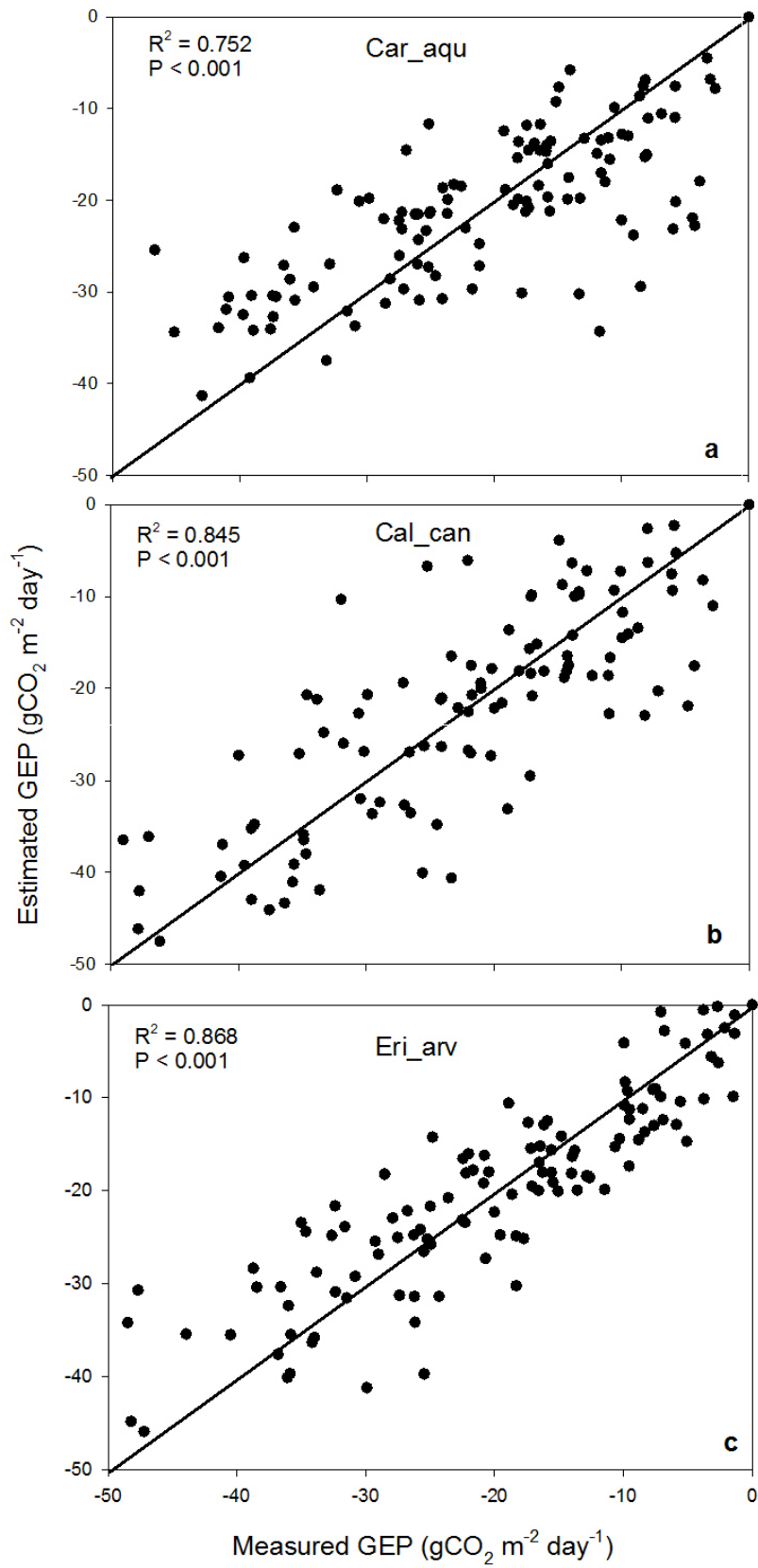
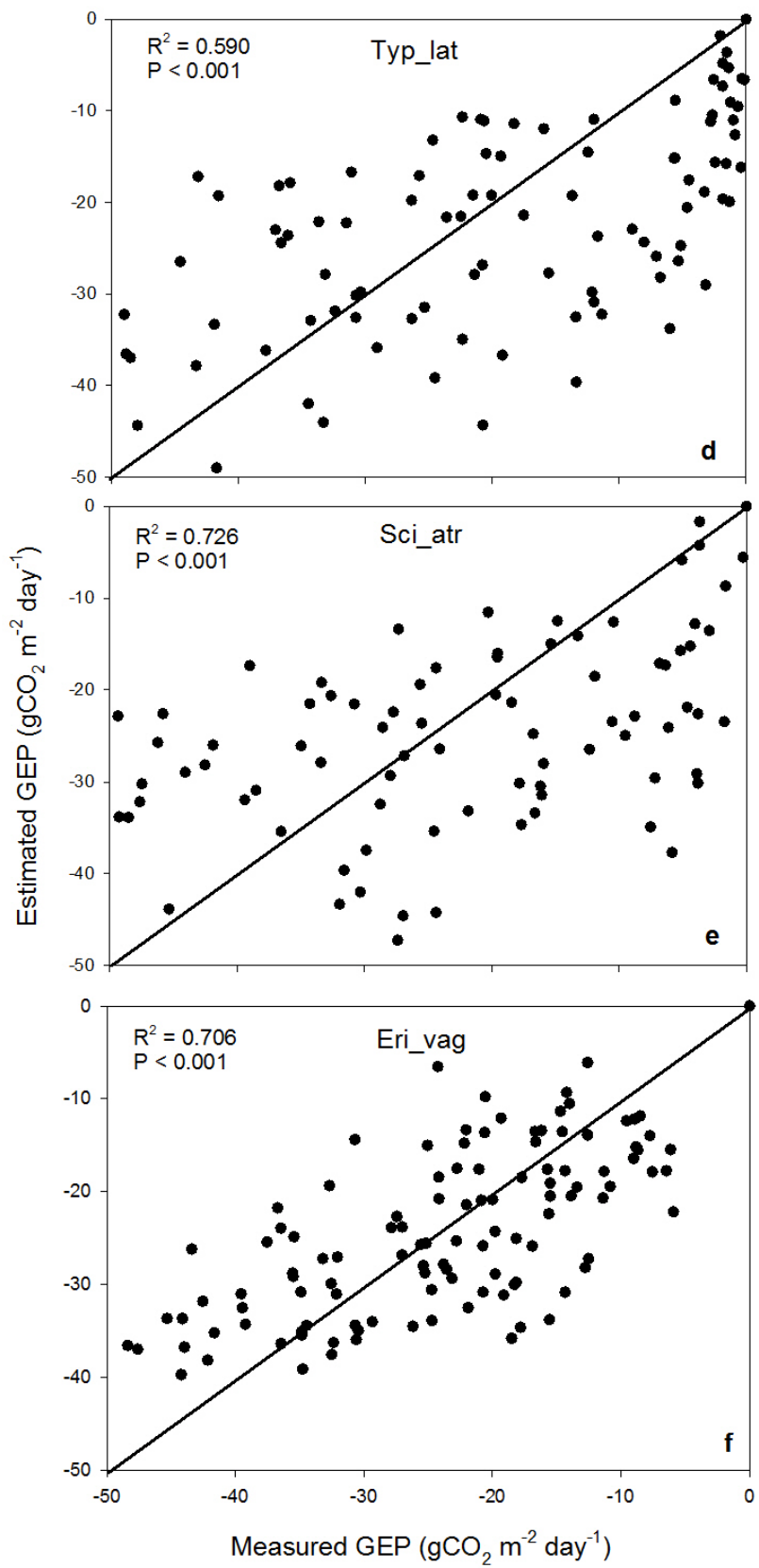


Figure – B.4: The distribution of all measured NEE_{max} in 2009 (a) and 2008 (b). Any positive value is showing source of CO_2 and negative showing sink of CO_2 .





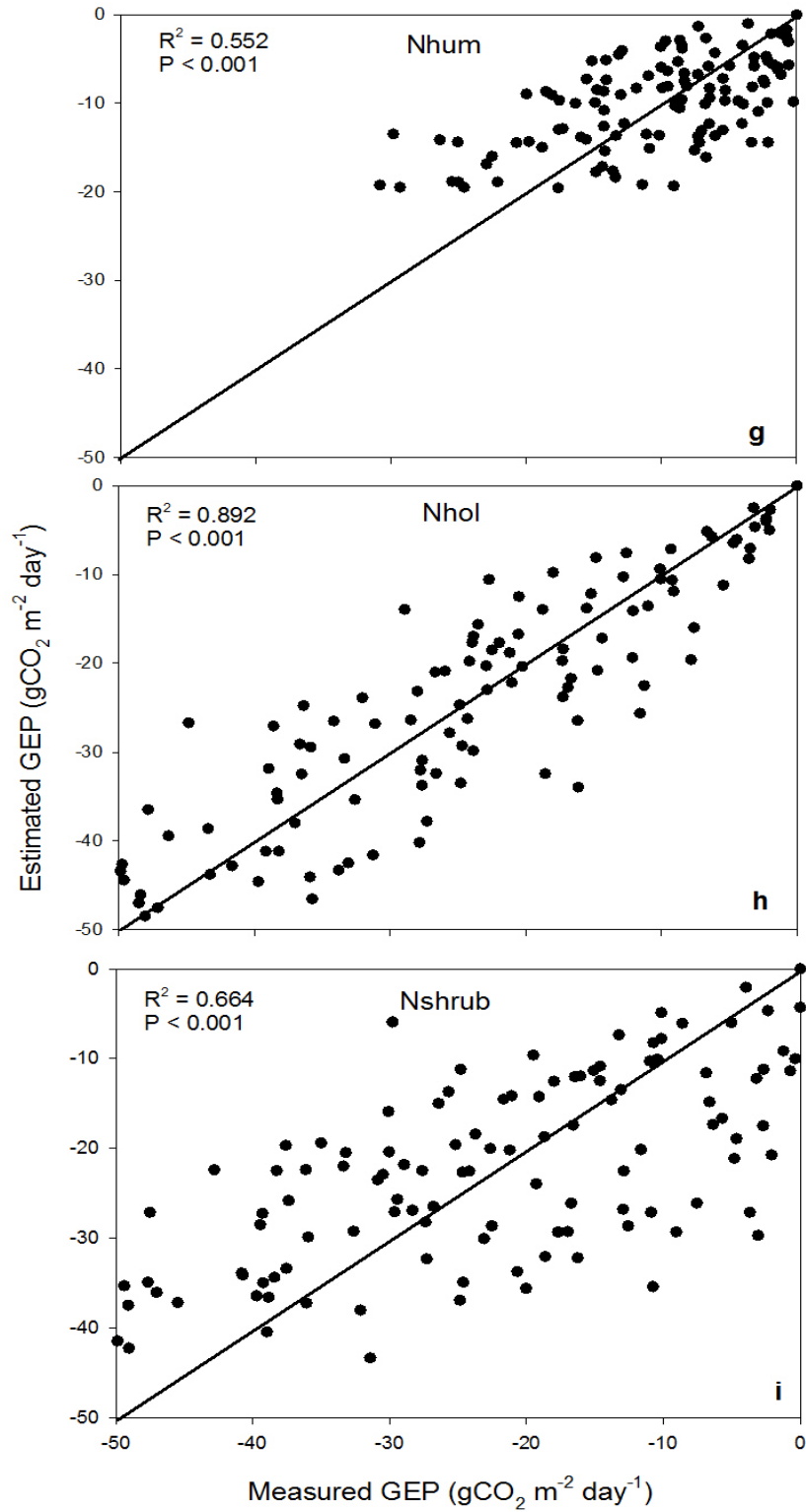
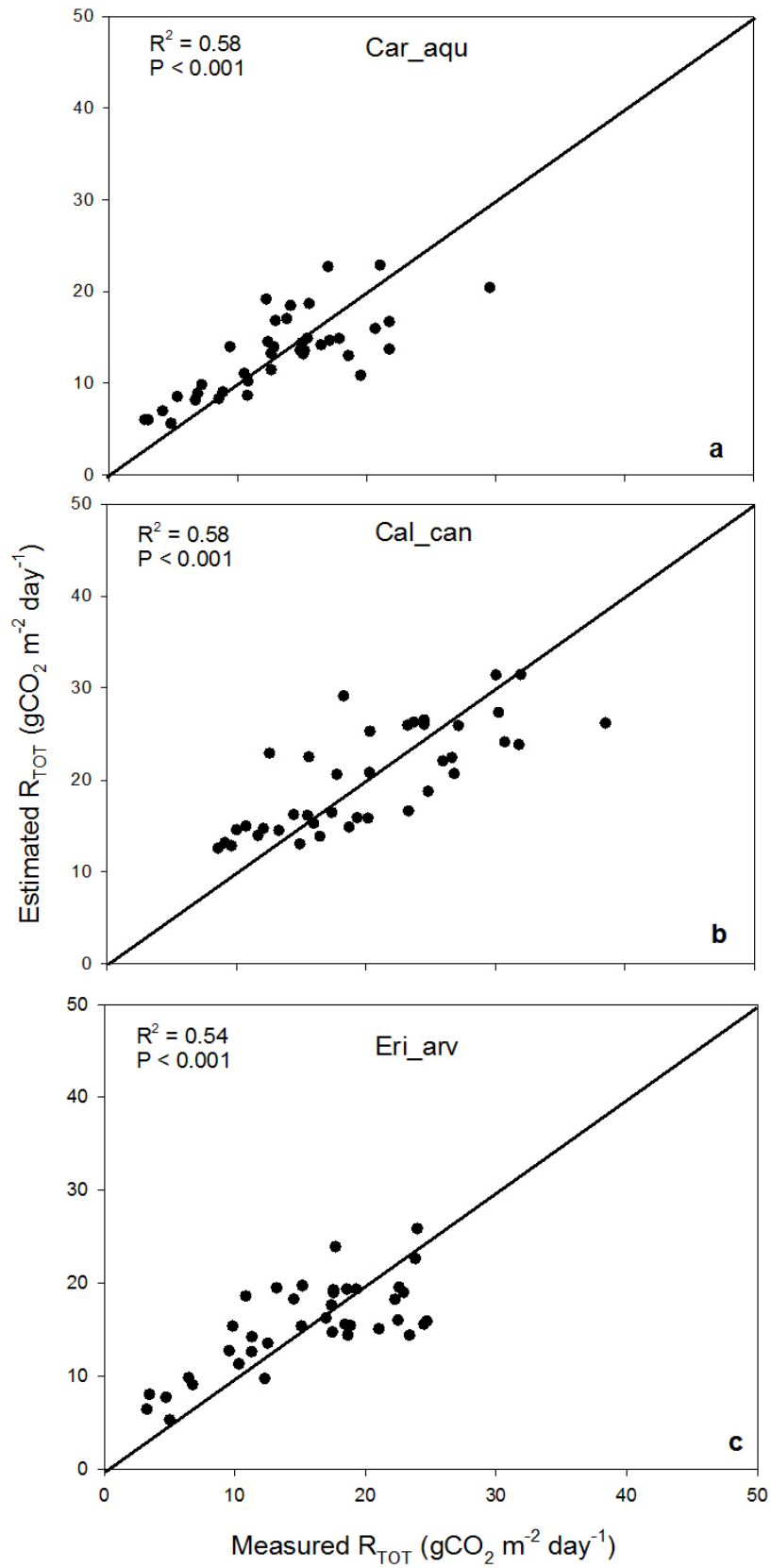
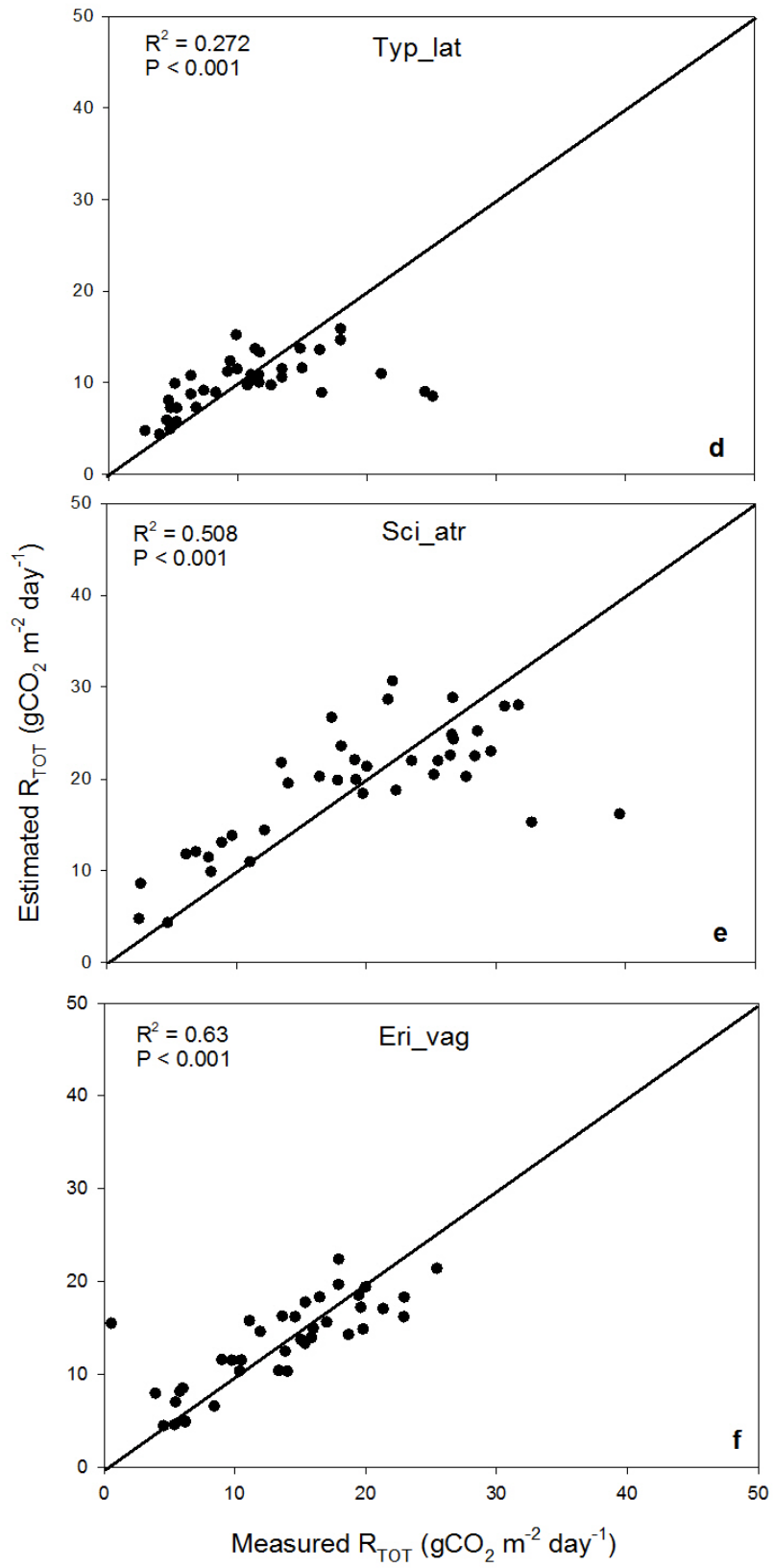


Figure – B.5: Comparison of measured and modelled GEP values for all vegetation types (a-i) (equation – 2.2). The 1:1 line shows the perfect match.





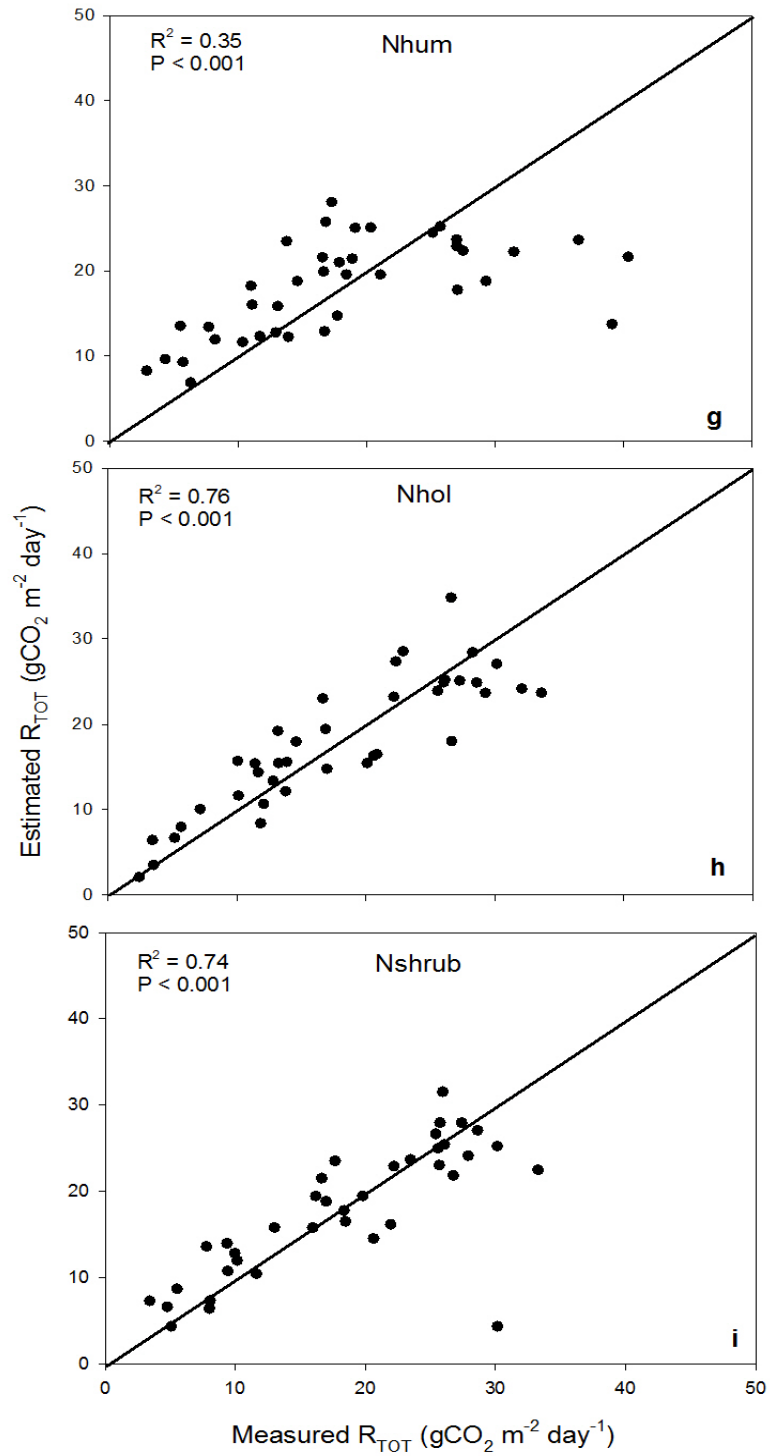


Figure – B.6: Comparison of measured and modelled R_{TOT} values for all vegetation types (a-i) (equation – 2.2). The 1:1 line shows the perfect match.